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Canadian Water Quality Guidelines for Trifluralin

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Abstract

A literature review was conducted on the uses, fate, and effects of trifluralin on raw water for drinking water supply, freshwater aquatic life, agricultural uses, recreational water quality and aesthetics, and industrial water supplies. The information is summarized in this publication. From it, water quality guidelines for the protection of specific water uses are recommended.

Résumé

On a examiné la documentation relative à utilisation, au devenir et aux effets de la trifluraline sur l'eau brute utilisée comme eau potable, sur la vie aquatique en eau douce, sur l'utilisation de l'eau pour l'agriculture, sur la qualité et les aspects esthétiques des eaux récréatives, ainsi que sur les approvisionnements en eau industrielle. Ces informations sont résumées dans cette publication. À partir de celles-ci, on recommande des concentrations limites de trifluraline afin de protéger ces diverses utilisations de l'eau.

Canadian Water Quality Guidelines for Trifluralin

R.A. Kent, M. Taché, P.-Y. Caux, and B.D. Pauli

SOURCES, OCCURRENCE, AND CHARACTERISTICS

Uses and Production

Trifluralin, the common name for α,α,α-trifluoro-2,6-dinitro-*N*,*N*-dipropyl-*p*-toluidine (IUPAC) or 2,6-*N*,*N*-dipropyl-4-trifluoromethylaniline (CAS) is an orange crystalline solid compound with a molecular formula of $C_{13}H_{16}F_3N_3O_4$ and a molecular weight of 335.5 (Worthing and Walker 1987). The Chemical Abstracts Service (CAS) registry number is 1582-09-8. Trade names for trifluralin and its various formulations registered in Canada include Treflan, Triflurex, Co-op Garden Weed Preventer, Heritage Selective Granular Herbicide, Rival, and Fortress (Agriculture Canada 1989).

Trifluralin is available as 400, 450, 500, and 545 $g \cdot L^{-1}$ active ingredient (ai) emulsifiable concentrates, 1.47%, 4%, 5%, and 10% ai granules, and 95%–96% ai technical product (Agriculture Canada 1989). It is used to control a wide range of annual grasses and broadleaf weeds in soybeans, dry beans (white or kidney), faba beans, snapbeans, lima beans, black beans, canola (rapeseed), triazine-tolerant canola, sunflowers, turnips, peas (field and canning), and direct seeded alfalfa; transplants of tomatoes, peppers, brussels sprouts, broccoli, cabbage, and cauliflower; carrots, crambe, direct seeded cabbage and cauliflower, annual flowers, woody nursery stock, perennials, and established shelterbelts (Ontario Ministry of Agriculture and Food 1989). Trifluralin is usually preplant incorporated because of its volatility (Maguire *et al.* 1988) and has very little activity after emergence (Worthing and Walker 1987). For effective result, soil incorporation concentrations may range from 0.5 to 1.0 $kg\text{-}ai \cdot ha^{-1}$ (Worthing and Walker 1987).

Trifluralin is not manufactured in Canada and was first registered in 1965 (Agriculture Canada 1989). Reported imports of trifluralin-formulated herbicides and other pesticide and non-pesticide toluidine isomer derivatives amounted to 7542, 3560, 6621, and 4801 t for 1984, 1985, 1986, and 1987, respectively (Statistics Canada 1988). These quantities, however, refer to the mass of the formulated product (which includes the active ingredient) and likely contain solvents and additives (e.g., surfactants); the formulations may also consist of secondary pesticide active ingredients. In addition, there are often several categories under which a product could potentially be classified. Therefore, a single category (e.g., formulated herbicides) may not reflect the total importation of a particular pesticide (Statistics Canada 1988).

Physical and Chemical Characteristics

The structural formula for trifluralin is presented in Figure 1. Selected physical and chemical properties of trifluralin are presented in Table 1. There appears to be wide disagreement on trifluralin water solubility in the literature with reported values ranging from 0.05 to 4 $mg \cdot L^{-1}$. This variation could be due to the use of different temperatures and to the method by which the values were generated, i.e., whether these were measured or calculated. Discrepancies in the values obtained for the sediment/water distribution coefficient (K_{oc}) may be due to the units of the K_d values that were used to calculate K_{oc} (B.T. Bowman 1990, Agriculture Canada, London, Ont., pers. com.). $K_{oc} = K_d/F_{oc}$ where K_d is the soil/water partition coefficient and F_{oc} is the organic carbon fraction. Whether K_d was derived from linear adsorption isotherms or from the Freundlich isotherm can significantly alter the magnitude of the K_d (B.T. Bowman 1990, Agriculture Canada, London, Ont., pers. com.).

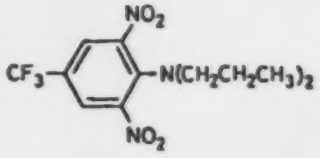

Figure 1. Structural formula for trifluralin.

Table 1. Physical and Chemical Properties of Trifluralin

Chemical formula	$C_{13}H_{16}F_3N_3O_4$[1]
Molecular weight	335.5[1]
Physical state	Orange, crystalline solid[1]
Henry's law constant	4.02 Pa $m^3 mol^{-1}$ (at 20°C)[2]
Melting point	48.5°C–49°C[3] 46°C–47°C[2]
Boiling point	139°C–140°C at 4.2 mm Hg[3]
Vapour pressure	2.93×10^{-3} Pa at 25°C[7] 2.62×10^{-2} Pa at 29.5°C[3] 6.46×10^{-3} Pa at 20°C[2] 3.23×10^{-2} Pa at 30°C[2]
Octanol/water partition coefficient log (K_{ow})	1149[2] 66007[6] 112030[6] 218080[6] 190060[7]
Sediment/water distribution coefficient log (K_{oc})	13700[7] 7340[7] 17750[7]
Solubility: water	4 mg·L^{-1} at 27°C[3] 1 mg·L^{-1} (temp. NR)[4] 0.05 mg·L^{-1} (temp. NR)[5,7] <1 mg·L^{-1} at 27°C[4] 0.3 mg·L^{-1} (temp. NR)[8]
acetone	400 g·L^{-1}[3]
xylene	580 g·L^{-1}[3]
Elemental analysis	C, 46.57%; H, 4.81%; F, 17.00%; N, 12.53%; O, 19.09%[4]
Half-life in topsoil	19–450 d[9]

[1]Worthing and Walker, 1987. [2]Suntio *et al.* 1988. [3]Verschueren 1983. [4]Windholz *et al.* 1983. [5]Poe *et al.* 1988. [6]Newsome, Lipnick, and Johnson 1984. [7]Huckins, Petty, and England 1986. [8]Grover, Smith, *et al.* 1988. [9]U.S. EPA 1984.

NR = not reported

Methods of Analysis

The method of analysis for trifluralin is by gas liquid chromatography (GLC) or by colorimetry; residues are determined by GLC with an electron capture detector (ECD) (Worthing and Walker 1987). Detection limits are in the order of 0.003 µg·L^{-1} in surface waters. Air, soil, water, and biota analytical procedures for trifluralin residues are available (Grover and Kerr 1981; Grover, Smith, *et al.* 1988; Lee and Chau 1983; Spacie and Hamelink 1979; Camper, Stralka, and Skipper 1989).

Mode of Action

Trifluralin appears to act as a mitotic poison affecting root growth (Ashton and Crafts 1973; Probst, Golab, and Wright 1975; Poe *et al.* 1988). Various cytological studies have established that trifluralin arrests mitosis by inhibiting the polymerization of tubulin, which is necessary for the formation of microtubules in the mitotic spindle apparatus (Vaughn and Vaughn 1986; DiTomaso 1988). The concentrations reported for disruption of the mitotic sequence is in the range of 1–4 µM (335.5–1342 µg·L^{-1}) trifluralin for root meristems of wheat (*Triticum aestivum*) (Lignowski and Scott 1972) and 0.3 µM (100 µg·L^{-1}) for the liquid endosperm of an African blood lily (Jackson and Stetler 1973). As trifluralin is primarily used as a soil-incorporated herbicide, its target is the plant root system, especially the lateral roots (Bayer *et al.* 1967; Mitchell and Bourland 1986; Cranfill and Rhodes 1987; Sparchez *et al.* 1987). Trifluralin may also affect other metabolic reactions, such as lipid synthesis, on a plant-specific basis (Sparchez *et al.* 1987).

At the molecular level, trifluralin is known to inhibit chloroplast electron transport reactions (Moreland *et al.* 1969). Further investigations into the macromolecular effects of trifluralin indicate that its effect on electron transport may be the result of the partitioning of the trifluralin molecule into the inner membrane lipid and the subsequent alteration of membrane fluidity, alignment of electron transport components, and lipid bilayer modulation (Moreland and Huber 1979). Other studies indicate that trifluralin inhibits energy-dependent calcium uptake in plant mitochondria at concentrations less than those interfering with tubulin polymerization (Hertel and Marmé 1983).

Entry into the Environment

After application to soil, trifluralin has the potential to leave the site and disperse in the environment. Because it is a relatively volatile compound, volatilization is a major dissipation pathway. Trifluralin has been found to occur in the air at concentrations as high as 160 ng·m^{-3} in regions of Canada where it is extensively used. The occurrence of trifluralin in air generally follows the seasonal use patterns for this herbicide, although soil moisture and rainfall events can also influence the timing and concentrations in the air (Grover, Kerr, *et al.* 1988).

Another transport pathway allowing trifluralin dispersion in the environment is surface water runoff from treated fields (Willis, Rogers, and Southwick 1975). A summary of trifluralin concentrations in runoff water from treated fields is presented in Table A-1. Trifluralin is expected to exhibit minimal movement

from soils because of its low solubility and strong adsorption to soil (Helling and Turner 1968; Helling 1971), and as a result, low concentrations are found in runoff water. This is generally true as trifluralin concentrations in runoff from treated fields typically range from below detection limits (approximately. 0.01 $\mu g \cdot L^{-1}$) to 0.04 $mg \cdot L^{-1}$ in bulk water samples (Axe, Mathers, and Weise 1969; Sheets, Bradley, and Jackson 1973; Willis, Rogers, and Southwick 1975; Wauchope 1978; Leonard, Langdale, and Fleming 1979; Rhode *et al.* 1980; Grover 1983; Willis *et al.* 1983). Trifluralin has been detected in runoff for as long as 12 months after application (Wauchope 1978).

Sediment is considered to be the primary transport vector for trifluralin as a result of erosion from treated fields. Although trifluralin and sediment yields have been found to be poorly correlated ($r = 0.48$) over several years, better correlations exist between these two variables when considering the data for individual storm events or on an annual basis ($r = 0.74–0.99$) (Willis *et al.* 1983). As much as 84% of the trifluralin in runoff may occur in the sediment phase (Sheets, Bradley, and Jackson 1973; Wauchope, Savage, and Chandler 1977). One report stated, however, that only 15% of the total trifluralin occurred in the sediment phase (Leonard, Langdale, and Fleming 1979). Although herbicide concentrations, in general, can be 2–3 orders of magnitude higher in sediments than in the sediment-associated water, because sediment is usually a small fraction of the total runoff, most of the herbicide loss occurs in the runoff water (Wauchope 1978).

Trifluralin concentrations in runoff water are decreased by incorporation of the herbicide into soil (Leonard, Langdale, and Fleming 1979). Trifluralin concentrations in runoff are usually highest after the first post-application rainfall or irrigation event and decrease afterward (Grover 1983). The decrease of trifluralin in runoff water with time, described as both gradual (Sheets, Bradley, and Jackson 1973) and exponential (Leonard, Langdale, and Fleming 1979), reflects the decrease in herbicide concentration in the runoff-active zone at the soil surface. Trifluralin concentrations in subsurface or phreatic runoff have been reported to contain concentrations of trifluralin 2 orders of magnitude lower than the corresponding surface runoff (Rhode *et al.* 1980).

Factors such as the slope of the land, soil, and rainfall apparently all contribute to the final concentration of trifluralin in runoff water (Leonard, Langdale, and Fleming 1979). Edge-of-field trifluralin concentrations in runoff are reduced by dilution in receiving waters and by adsorption to stream sediments, untreated soils, or vegetation surfaces (Wauchope 1978; Grover 1983). Crop type does not appear to affect the runoff concentration of trifluralin (Willis, Rogers, and Southwick 1975), however, the tillage practice does, as shall be discussed later.

Environmental Concentrations

A summary of trifluralin concentrations in Canadian surface waters, groundwater, sediment, and biota is presented in Table B-1. Trifluralin concentrations in streams in areas where the herbicide is used range from 0 to 1.8 $\mu g \cdot L^{-1}$ and are frequently below detection limits. The lowest detection limit reported was 3 $ng \cdot L^{-1}$ (Williamson 1984; Waite *et al.* 1986; Muir and Grift 1987; Therrien-Richards and Williamson 1987).

The concentration of trifluralin in surface waters has been found to follow a biannual pattern: increases occur during spring runoff and then again during autumn rains when erosion is expected to be greatest, particularly on the prairies (Williamson 1984). Increased concentrations in surface waters, however, are also reported to result from the deposition of trifluralin vapours or dust particles with adsorbed trifluralin from neighbouring applications (Muir and Grift 1987).

Nondetectable surface water residues in areas of the country during the normal spring application period were suggested to be the result of lower than normal precipitation (Therrien-Richards and Williamson 1987).

Because of the very low water solubility of trifluralin, contamination of groundwater is not expected to occur by leaching within the soil column. Contamination is more likely to occur by direct deposition or surface water runoff into wells. A survey of 91 farm wells across southern Ontario found trifluralin contamination at 41 $\mu g \cdot L^{-1}$ in one well. This well was 1 of 14 from which water was drawn for pesticide formulating and spraying, and contamination was thought to have occurred during filling of the spray tanks (Frank *et al.* 1987).

The U.S. national water quality monitoring data base, STORET, contains trifluralin monitoring data from 511 selected sampling sites. Trifluralin concentrations in surface water ranged from 0.1 to 51 $\mu g \cdot L^{-1}$ (U.S. EPA 1984). The number of nondetectable versus

detectable trifluralin residues in the survey was not provided.

Slightly soluble pesticides such as trifluralin tend to be readily adsorbed and accumulated on bottom sediment and particulate matter (Therrien-Richards and Williamson 1987). Detectable levels of trifluralin, however, have been found in very few sediment samples. When detected, concentrations are typically higher than those in water on a weight or volume basis. Sediment trifluralin concentrations generally range from 4 to 6 $\mu g \cdot kg^{-1}$ (Therrien-Richards and Williamson 1987). Sediment concentrations of trifluralin in the United States ranged from 4 to 5000 $\mu g \cdot L^{-1}$ with an average of 115 $\mu g \cdot L^{-1}$ (U.S. EPA 1984).

Environmental Fate, Persistence, and Degradation

Fate, persistence, and degradation are highly dependent on the peculiarities of chemical molecules. The dinitro functional group of trifluralin and other dinitroaniline herbicides extensively decreases the molecules' water solubilities as these make hydrogen bonds with alkyl groups of surrounding molecules. This has the effect of forming lipophilic micelles resisting solvation into water (Weber 1987). It has been suggested that the binding of dinitroanilines is due to hydrogen bonds between the nitro groups and proteinaceous sites in soil organic matter and/or charge transfer bonds between high charge density aromatic rings in soil humic substances and the low charge density aromatic rings of the dinitroanilines (Weber 1987).

Soil

The persistence of trifluralin in soil depends on the biotic and abiotic characteristics of the soil and the behaviour of the herbicide in the soil. A summary of selected soil persistence studies is presented in Table C-1. Investigations in Canada and the United States have demonstrated that the data vary widely due to the uniqueness of the locations and of the experimental conditions used by the different analysts. Generally, cool, dry climates favour greater persistence than warmer, more moist conditions (Jensen, Ivany, and Kimball 1983; Weber 1990). Decreased persistence has been observed with increasing temperature (Horowitz, Hulin, and Blumenfeld 1974; Smith 1975; Duseja 1982) and increasing available moisture (Smith 1975; Savage 1978; Duseja 1982; Pchajek, Morrison, and Webster 1983). The absence of a soil microbial community also appeared to increase persistence in soils (Mostafa *et al.* 1982).

Trifluralin soil half-life values for Nova Scotia and Prince Edward Island are reported to range from 140 to 164 d in sandy loam soils (Jensen, Ivany, and Kimball 1983) and can be as low as 63 to 77 d for southwestern Ontario (Gaynor 1985). Within regions of similar climate, seasonal differences would be expected to influence persistence. Trifluralin applications in the fall would be expected to persist longer than spring applications (Pchajek, Morrison, and Webster 1983) due to the supression of biological and chemical activity in frozen ground. Overwinter losses of trifluralin applied in the fall, however, may be as high as 38% (Jensen and Kimball 1980).

Soil residue carry-over has been observed to be as high as 47% in Nova Scotia and 38% in Saskatchewan after 1 year (Smith and Hayden 1976; Jensen and Kimball 1980) and 16% in Saskatchewan after 1.5 years (Smith 1975; Smith and Hayden 1976). Carry-over of phytotoxic levels (concentrations not reported) after 1 or more years has been reported in Ontario (Gaynor 1985). In the cooler climate of Alaska, 26% to 51% of trifluralin applied in the spring at 1.1 $kg \cdot ha^{-1}$ was found at the end of the growing season. Only a small quantity (10%) of that found at the end of the growing season was lost during the winter. Differences in trifluralin persistence in the Alaskan studies were not related to the application rate or soil type (Conn and Knight 1984).

Studies conducted in the warmer climates of Tennessee (Duseja, Akunuri, and Holmes 1980) and Texas (Menges and Hubbard 1970) have reported half-year values of 1%–3%. Little evidence for accumulation of trifluralin in U.S. soils, even after repeated applications, has been reported (Parka and Tepe 1969; Burnside 1974; Miller *et al.* 1978).

Trifluralin is usually applied preplant and soil incorporated and is strongly adsorbed to soil particles, especially in those soils with high organic matter content (Eshel and Warren 1966; Webster *et al.* 1978; Grover, Banting, and Morse 1979; Bush, Abernathy, and Gipson 1982). Increasing persistence has been reported to be associated with increased soil organic matter (Bardsley, Savage, and Childers 1967). This correlation, however, is variable. Weber (1990) stated that this variation may be due to the high variation in the literature-reported K_{oc} values. Soils having organic matter contents below about 4% have been reported to show an increase in trifluralin persistence or a lack of any relationship (Smith 1975; Smith and Hayden 1976; Duseja and Holmes 1978; Duseja, Akuniri, and

Holmes 1980; Jensen and Kimball 1980; Solbakken *et al.* 1982; Jensen, Ivany, and Kimball 1983). Above approximately 4.0% organic matter, Smith (1975) and Smith and Hayden (1976) found increasing trifluralin persistence with increasing organic matter. Similar results with soils increasing in organic matter from 2.9% to 8.8% were reported by Webster *et al.* (1978). The evidence from the compiled literature supports a positive relationship between soil organic matter and persistence when soil organic matter exceeds 4%. Experiments with various adsorbents ranging from kaolinite and montmorillonite clays to activated charcoal demonstrated strong trifluralin adsorption onto hydrophobic adsorbents such as charcoal, peat moss, and cellulose triacetate (Grover 1974). Trifluralin adsorption to clays is weak and it is easily desorbed in the presence of water. The non-ionic nature of trifluralin suggests that soil pH would have minimal effects on adsorption (Hollist and Foy 1971).

The trifluralin pattern of degradation has been described as typical of first-order rate kinetics (Zimdahl and Gwynn 1977; Webster *et al.* 1978; Golab, Althaus, and Wooten 1979; Duseja, Akunuri, and Holmes 1980; Gaynor 1985; Smith, Aubin, and Derksen 1988). Strict first-order rate kinetics, however, tends to underestimate the initial rapid degradation phase of trifluralin in soil and overestimate the second, slower phase (Reyes and Zimdahl 1989). Field studies of trifluralin degradation at four locations in Colorado in June produced data, which, when described mathematically, produced a biexponential equation. This equation resulted from the integration of first-order and second-order differential rate equations and described the observed field degradation of trifluralin at 15 of 25 soil-site combinations better than strict first-order rate kinetics (Reyes and Zimdahl 1989).

Large-scale field trials in southern Saskatchewan demonstrated three distinct phases in the dissipation of trifluralin from soil. Initially there is a rapid dissipation phase, lasting about 1 week, with vapour loss being the major route, especially under moist soil conditions. A second, slow dissipation phase, lasting over the entire growing season, occurs as the result of a combination of volatilization, adsorption, and microbial degradation. The third or no dissipation/breakdown phase follows soil freezing in the fall and lasts to the spring thaw under typical Canadian prairie conditions. Gross dissipation of trifluralin during phases one and two follows a first-order rate of reaction with a half-life of approximately 99 d (Grover *et al.* 1988).

The various processes governing the persistence and fate of trifluralin in the environment include volatilization, photodegradation, and microbial degradation. Chemical hydrolysis is not considered important in the fate of trifluralin as it is stable at pH 3, 6, and 9 (U.S. EPA 1987a).

Volatilization

Volatilization can be an extremely important factor in the loss of trifluralin from soil. Under some circumstances, the vapours leaving the application site can be of sufficient concentration to kill or injure nearby seedlings (Swann and Behrens 1972a, 1972b). The rate of loss depends on the amount of trifluralin applied to the surface of a moist soil and decreases in proportion to the inverse square root of the hours of daylight post-application (Glotfelty *et al.* 1984). This relationship exists because soil temperature, as controlled by solar insolation, is a major factor in the diffusion-controlled volatilization of trifluralin from moist soils. The report by Roggenbuck and Penner (1987) concluded that at 15°C, an application of 0.22 $kg \cdot ha^{-1}$ trifluralin reduced shoot and root fresh and dry weights and shoot length in corn (*Zea mays*) seedlings to a much greater extent than at 25°C. The increased temperature apparently allowed extensive volatilization to occur thus reducing seedling injury. Volatilization of surface-applied trifluralin may be extensive. Laboratory chamber studies of volatilization showed rates of 1.15 $kg \cdot ha^{-1} \cdot d^{-1}$ for trifluralin applied at 2.5 $kg \cdot ha^{-1}$ to bare soil at 35°C. Moisture was maintained in the soil for the 154-d test period. Volatilization increased 1.8 times for each 10°C temperature increase (Nash and Gish 1989). During a field study, 2.84 $kg \cdot ha^{-1}$ (4.7 $kg \cdot ha^{-1} \cdot d^{-1}$) volatilized from a moist soil with a temperature of 19°C and a wind speed at 1 m height of 5 $m \cdot s^{-1}$ during the period immediately following the application of 2.8 $kg \cdot ha^{-1}$ trifluralin (Glotfelty *et al.* 1984).

Soil organic matter is also a factor in trifluralin volatilization; stronger adsorption restricts vapour loss. In a laboratory study, vapour densities of 3.19, 1.73, and 0.62 $\mu g \cdot L^{-1}$ corresponded to soil organic matter contents of 0.20%, 0.58%, and 1.62%, respectively (Spencer and Cliath 1974). Soil moisture itself, however, is probably the major factor in trifluralin volatilization (Ketchersid, Bovey, and Merkle 1969; Harper *et al.* 1976; Grover 1983). Moist soil allows trifluralin vapour loss as the soil moisture competes with trifluralin for adsorption sites on the organic matter fraction. Thus, relatively small amounts of

moisture, such as that present in dew, may greatly enhance trifluralin volatilization (Glotfelty *et al.* 1984).

Air samples collected at Regina, Saskatchewan, at six heights (ranging from 30 to 200 cm) above the soil surface after trifluralin (0.74 $kg \cdot a^{-1}$) incorporation into the soil (5 cm) and then above the crop canopy following emergence 67 d after application, showed distinct gradients of trifluralin vapours in the air. The highest trifluralin air concentrations occurred closest to the ground. The highest flux rate for trifluralin was 3 $g \cdot ha^{-1} \cdot h^{-1}$ during the 4- to 6-h period after application when the concentration at 30 cm above ground was 1700 $ng \cdot m^{-3}$. The flux of trifluralin decreased with time and was dependent on soil moisture conditions. Total trifluralin vapour loss from the 67-d period was 23.7% (Grover *et al.* 1988).

Incorporation of trifluralin into the soil may retard, but will not eliminate, loss by volatilization. A 21-d half-life was reported for trifluralin incorporated into 2.5 cm in a Texas soil. Surface application to soil, without incorporation, can lower the half-life to between 1 and 18 h due to volatilization (Glotfelty *et al.* 1984) and possibly photodegradation. A trifluralin application of 1.2 $kg \cdot ha^{-1}$ incorporated to a depth of 7.5 cm decreased volatilization loss to 1.65% compared to 10.7% for that applied to the surface of the soil (Oliver 1979). Other studies that appear to show reduced volatilization with soil incorporation are complicated by differences in experimental conditions. A 2.5-cm incorportion depth resulted in a 22% volatilization loss in 120 d (White *et al.* 1977). The soil used in this study was a Georgia sandy loam with low (0.55%) organic matter content. By contrast, trifluralin incorporated to 7.5 cm in a heavier textured New York loam soil with 3%–4% organic matter produced only 3.4% volatilization loss in 90 d (Taylor 1978). The differences in soils and weather between the two experiments, however, account for the marked differences in volatilization losses, not the depth of incorporation (Taylor 1978).

Conventional tillage practices using a moldboard plow may "dilute" trifluralin concentrations if the depth of disturbance is below the trifluralin incorporation depth thus reducing persistence (Hartzler, Fawcett, and Owen 1989).

An attempt to simulate the environmental partitioning and fate of ^{14}C-trifluralin was conducted by applying the herbicide as a foliar spray to a terrestrial microcosm chamber at 0.28 $kg \cdot ha^{-1}$. Trifluralin is not normally applied to foliage because of its high volatility and susceptibility to photodecomposition. Accordingly, 62% of the total applied radioactivity in the air of the microcosm was found after 19 d. The plants in the microcosm accounted for 21% of the residual radioactivity while the soil contained 15% (Gile, Collins, and Gillet 1980).

Photolysis

Once released into the atmosphere, trifluralin is known to undergo vapour phase photochemical transformation. These transformations have been studied in the laboratory and confirmed by field sampling. One pathway results in the N-dealkylation of one or both N-propyl groups ultimately ending in 2,6-dinitro-α,α,α-trifluoro-*p*-toluidine. A second pathway involves the internal condensation between one of the N-propyl side chains and one of the nitro groups. Final dealkylation of the other N-propyl side chain produces 2-ethyl-7-nitro-5-trifluoromethyl benzimidazole (Moilanen and Crosby 1975). In addition, dimerization of photochemically transformed trifluralin molecules and the subsequent photodegradation of the dimer produces at least two types of azobenzene and three types of azoxybenzene derivatives (Sullivan, Knoche, and Markle 1980).

The rates of photochemical transformation appear to be rapid, at least in the initial stages. The half-life for the photodegradation of vapour-phase trifluralin to a dealkylated product was 20 min during summer field studies (relative humidity 20%–30%, air temperature 20°C–30°C) (Woodrow *et al.* 1978). This half-life increased to 193 min in the fall under similar conditions of relative humidity and air temperature, but reduced daylight. These photochemical reaction rates are consistent with results obtained in laboratory studies (Woodrow *et al.* 1978). Other photolysis chamber studies derived half-lives ranging from 19 to 74 min in natural sunlight (Mongar and Miller 1988).

While these studies discuss vapour phase transformations, which might be expected in the air or on the surface of vegetation, little is known concerning the potential for trifluralin photodegradation on soil. One study reported that photodecomposition of trifluralin on soil particles did not occur during 9 h of exposure to natural sunlight (Plimmer 1978). Trifluralin exposed to sunlight on a soil surface at 0.07–0.28 $kg \cdot ha^{-1}$ for a period of 2 h had substantially reduced herbicidal activity measured by bioassay in comparison to unexposed trifluralin. Activity was reduced further with longer exposures, but the rate at which

herbicidal activity decreased slowed (Wright and Warren 1965). Dry soil thin layer plates, where trifluralin was applied at 1 kg·ha^{-1} and exposed to natural sunlight for 7 d, exhibited a loss of 18.4% compared to dark controls (Parochetti and Dec 1978).

The similarity of trifluralin photoproducts from soil suspensions and in water alone suggests that trifluralin photodecomposition in moist soils occurs in homogenous solution with the rate dependent upon the equilibrium between soil adsorbed and dissolved trifluralin (Crosby and Leitis 1973).

Microbial Degradation

Trifluralin is degraded by soil microorganisms *via* aerobic and anaerobic pathways. Aerobic biodegradation usually involves a series of oxidative dealkylation steps, whereas anaerobic conditions generally result in the reduction of the nitro groups. Both biodegradation systems may occur in the same field soil (Camper, Stralka, and Skipper 1980; Zeyer and Kearney 1983). Relative rates of biodegradation are dependent on environmental moisture and oxygen conditions and are greatest in moist anaerobic conditions followed by flooded anaerobic and finally moist aerobic (Parr and Smith 1973; Junk, Richard, and Dahim 1984).

Laboratory biodegradation studies have demonstrated the potential for soil microbes to degrade trifluralin. Rapid biodegradation of tritiated trifluralin by pure cultures of three fungal species common in Egyptian soil was reported by Zayed *et al.* (1983). Approximately 91%–97% biodegradation was reported in the pure fungal cultures after 10 d at 25°C when incubated in the dark. It is noteworthy that the volatilization of trifluralin from the liquid culture media was not monitored. This may have contributed to the biodegradation.

Trifluralin containing ^{14}C-labelled propyl groups, ^{14}C-labelled ring carbons, or a ^{14}C-labelled CF_3 group was used by Zeyer and Kearney (1983) to monitor trifluralin biodegradation by pure strains and mixed cultures of soil microorganisms. Of the 180 strains of soil microorganisms tested, only 60 strains evolved $^{14}CO_2$ ranging from 1.5% to 11% of the added trifluralin within 21 d under dark, aerobic conditions at 26°C. The medium supporting the microbial growth contained carbon sources other than trifluralin. None of the 60 strains was able to grow with trifluralin as the sole carbon source. The amount of $^{14}CO_2$ evolved from mixed cultures never exceeded 1.6% of the trifluralin added. The slow liberation of $^{14}CO_2$ by the mixed microbial population probably resulted from slow degradation and high adsorption to particulates of the 50 mg·L^{-1} trifluralin added (Zeyer and Kearney 1983). The lack of substantial trifluralin biodegradation in soils is supported by laboratory studies of ^{14}C-trifluralin biodegradation in estuarine sediments. The release of $^{14}CO_2$ could not be identified from sediments placed in plastic cylinders and monitored over a 100-h period (Spain and van Veld 1983). By contrast, Means, Wijayaratne, and Boynton (1983) produced a half-life of 9 d for trifluralin in estuarine sediments in outdoor microcosms. Some of the conflicting results of trifluralin degradation in soils and sediments may be due to differences in the redox potential of the soil/sediments. This parameter substantially affects trifluralin degradation. At a redox potential of +150 mV, about 60% of trifluralin, initially present at 1 µg·g^{-1} in a soil suspension, remained after 21 d. Reducing the redox potential to +50 mV caused almost all the trifluralin to disappear in 8 d (Willis, Wander, and Southwick 1974). Similar results were obtained with trifluralin in a sediment slurry (Walker *et al.* 1988).

Extractable transformation products or metabolites of trifluralin have also been detected in soil at 4% of applied trifluralin levels, but evidence of their accumulation in soil was not found (Golab and Amundson 1975; Golab, Althaus, and Wooten 1979). As many as 28 identifiable and 4 unidentified metabolites appear to undergo further changes leading to complete mineralization. The total number of metabolites formed is not considered to be dependent on soil conditions. The metabolites have been proven to be less phytotoxic than the parent compound (Koskinen *et al.* 1986).

Proposed metabolic pathways for trifluralin biodegradation include mono- and di-dealkylation of N-alkyl substituents, reduction of nitro groups (the two major pathways), oxidation, hydrolysis, internal condensation, hydroxylation, dimeric condensation, and combinations of these processes (Golab and Amundson 1975; Golab, Althaus, and Wooten 1979; Camper, Stralka, and Skipper 1980; Mostafa *et al.* 1982; Zayed *et al.* 1983; Zeyer and Kearney 1983). As much as 50% of the total extractable metabolites is represented by polar condensation of aromatic amines which are formed by nitro group reduction (Mostafa *et al.* 1982; Zayed *et al.* 1983). Unlike the total number of metabolites, the number of these polar metabolites appears to be dependent on the soil texture and the number of soil microbes present during biodegradation

(Mostafa *et al.* 1982). Aromatic hydroxylation may aid in the cleavage of the ring eventually leading to the mineralization of these metabolites in the soil (Golab, Althaus, and Wooten 1979).

Trifluralin biodegradation also results in the formation of considerable quantities of soil-bound, nonextractable metabolites, which remain in the soil organic fraction (Golab and Amundson 1975). An extractable metabolite (α,α,α-trifluorotoluene-3,4,5-triamine) is considered to be a key metabolite in the formation of soil-bound, nonextractable residues (Golab, Althaus, and Wooten 1979). One year after trifluralin application, these residues were found to represent 43%–50% of the initial trifluralin levels (Golab and Amundson 1975; Golab, Althaus, and Wooten 1979). In another study, the nonextractable metabolites represented 45% and 72% of the originally applied trifluralin after 68 and 63 d, respectively (Wheeler *et al.* 1979). As much as 38% of the initial trifluralin application has been found to exist as soil-bound metabolites after 3 years (Golab and Amundson 1975; Wheeler *et al.* 1979).

Soil-bound residue concentrations are higher in soils with greater cation exchange capacity and percent organic carbon (% OC) (Wheeler *et al.* 1979). In a soil with higher organic matter content (3.87% OC), a strong relationship was found between the amount of binding and the substitution of the amino nitrogen of trifluralin and its metabolites (Wheeler *et al.* 1979). A reported higher percentage of soil-bound metabolites found in a sandy soil was attributed to the lower organic matter concentration and lower microbial density (Mostafa *et al.* 1982).

Water

Information concerning persistence of trifluralin in the aquatic ecosystem is mainly derived from microcosm studies. Trifluralin introduced as a single dose into artificial outdoor recirculating streams for a final concentration of 10 $mg \cdot L^{-1}$ caused no detectable changes in stream periphyton community structure and exhibited a half-life of 51 minutes (Kosinski 1984). Although this short half-life was attributed to photodecomposition, the results were inconclusive.

After introduction into wetland microcosms, ^{14}C-trifluralin disappearance from the water column approximated the biphasic sediment adsorption kinetics. Volatilization and photodegradation were identified as the major pathways for trifluralin removal from aquatic systems (Huckins, Petty, and England 1986). Volatilization of trifluralin from 310 mL of water (initial concentration <1 $mg \cdot L^{-1}$) in a laboratory chamber with an air flow of 20 L per minute at a temperature of 21°C–24°C was 100% after 24 h (Sanders and Seiber 1983).

A portion of the trifluralin lost from the water column to sediment adsorption is apparently returned to the water column in the form of more water-soluble degradation products (Huckins, Petty, and England 1986). Degradative mechanisms producing these more soluble degradation products were not discussed.

Karickhoff and Morris (1985) studied the effects of sediment-inhabiting oligochaete worms on trifluralin transport from the sediment to the water column using natural sediments (6–10 kg) equilibrated with 5–10 mg of trifluralin in approximately 20 L of distilled water. Approximately 1 kg of this trifluralin-containing sediment was transferred to microcosms with and without worms. The presence of worms dramatically altered the degradation of trifluralin producing rate constants of 0.2–0.4 d^{-1}. The cause of the enhanced trifluralin degradation was not specifically discussed.

The photolysis of trifluralin in natural waters depends on water depth, the magnitude and spectral distribution of sunlight, and the molar extinction coefficient of trifluralin. Trifluralin absorbs sunlight strongly in the visible region (390–800 nm), thus as water depth increases, the photolysis rate decreases. In northern latitudes, depth dependence of photolysis becomes more pronounced as the underwater path length of direct sunlight becomes longer as the sun is lower in the sky (Zepp and Cline 1977). Canadian waters would, therefore, have a higher potential for longer trifluralin half-lives given comparable conditions of water quality and volatilization than more southerly waters.

Photodecomposition of trifluralin in water follows pathways and provides photoproducts similar to those observed in the vapour phase studies. The presence of a photosensitizer (methanol) increased the photolytic reaction rate about ten times the rate observed in water alone. The presence or absence of soil (50 $g \cdot L^{-1}$) apparently had little effect on photolytic rate. Photodecomposition was rapid at acid pH, but the rate declined sharply and the proportions of the photoproducts changed above pH 7.4 (Crosby and Leitis 1973).

RATIONALE

Raw Water for Drinking Water Supply

Concentrations in Drinking Water

Published measurements of trifluralin in treated water in Canada were not found.

Removal by Water Treatment Operations

Treatment technologies for the removal of trifluralin from water are available and have been reported to be effective. Available data indicate that reverse osmosis, granular activated carbon adsorption, and conventional water treatment with alum will remove trifluralin from water. Selection of individual or combinations of technologies for trifluralin removal from water, however, must be based on a case-by-case technical evaluation (U.S. EPA 1987b).

Guideline

An interim maximum acceptable concentration (IMAC) for trifluralin in drinking water of 45 $\mu g \cdot L^{-1}$ has been proposed by Health and Welfare Canada (1987) as this herbicide is under review by this agency. The World Health Organization has established a drinking water quality guideline value of 170 $\mu g \cdot L^{-1}$ for trifluralin (WHO 1987).

The U.S. Environmental Protection Agency, Office of Drinking Water, issued a draft health advisory for trifluralin in August 1987. Health advisories describe nonregulatory concentrations of drinking water contaminants at which adverse health effects would not be anticipated to occur over specific exposure durations. Health advisories contain a margin of safety to protect sensitive members of the population. The 1-d, 10-d, 7-year, and lifetime exposure health advisories for trifluralin are 25, 25, 25, and 2 $\mu g \cdot L^{-1}$, respectively (U.S. EPA 1987b).

Freshwater Aquatic Life

Bioaccumulation

Observed or calculated bioconcentration factors (BCFs) for aquatic organisms are presented in Table D-1. Most of the data were obtained during microcosm studies in which ^{14}C-trifluralin was applied to determine quantities of trifluralin in both water and tissue after specific periods of time. Generally, the water concentration of trifluralin in these microcosm studies was not stable whether the trifluralin entered the water as a result of desorption from treated soil in a static system or was continuously input to the microcosm with an automated dilution apparatus. Mosquitofish (*Gambusia affinis*) BCFs ranged from 750 to 3140 for water containing 0.2–0.9 $\mu g \cdot L^{-1}$ trifluralin released from treated soil. Higher soil concentrations produced trifluralin water concentrations ranging from 3.4 to 9.1 $\mu g \cdot L^{-1}$ and from 36.9 to 160.1 $\mu g \cdot L^{-1}$ under static conditions. *G. affinis* BCFs under these conditions ranged from 300 to 1080 and from 70 to 1150, respectively (Yockim, Isensee, and Walker 1980). There seemed to be a tendency for the lower water concentrations to produce higher BCFs, but this was not discussed by the authors. Fish continually exposed to trifluralin doses for 30 d exhibited BCFs ranging from 1800 to 11 000 for water concentrations of 0.1–0.8 $\mu g \cdot L^{-1}$, from 2080 to 5710 for water concentrations of 0.5–2.6 $\mu g \cdot L^{-1}$, and from 1190 to 4050 for water concentrations of 9.3–29.8 $\mu g \cdot L^{-1}$ (Yockim, Isensee, and Walker 1980).

BCFs for the filamentous green alga *Oedogonium cardiacum* were generally in the 100 range for static microcosms regardless of water concentration. The same organism in continuously dosed microcosms had BCFs in the 1000 and 10 000 range. Snail BCFs in static microcosms generally ranged from 10 to 100, regardless of water concentrations. An increase of 1–2 orders of magnitude was observed in the microcosms receiving continuous trifluralin doses (Yockim, Isensee, and Walker 1980). These BCFs were based on the ratios of radioactive substances in tissues and water, and did not make the distinction between trifluralin and trifluralin metabolites.

A static microcosm system containing the same types of organisms was used by Kearney, Isensee, and Konston (1977). Trifluralin desorption from treated soil produced a 30-d average water concentration of 7.5 $\mu g \cdot L^{-1}$. BCFs for algae (276), snails (400), *Daphnia* (92), and fish (33) were also based on ^{14}C activity in tissues and water. The actual identity of the compounds was unknown. Examination of water and fish extracts at the end of the experiment showed that trifluralin was not present; the ^{14}C activity was entirely due to polar metabolites.

A somewhat more controlled laboratory method for determining trifluralin accumulation in fathead minnows

(Pimephales promelas) produced a BCF of 3261 for static conditions over a 40-h exposure to 20 $\mu g \cdot L^{-1}$ (Spacie and Hamelink 1979). Trifluralin concentrations in the water were measured over the exposure period. The uptake of trifluralin from water was linear with a rate constant of 755.98 d^{-1}. Transfer of fish to uncontaminated water resulted in first-order depuration with a rate constant of 0.23184 d^{-1}. Other BCF estimates given by Spacie and Hamelink (1979) for other fish species were 1030 (rainbow trout, *Salmo gairdneri*), 1294 (mosquitofish, *Gambusia affinis*), 5800 (sauger, *Stizostedion canadense*), 2800 (shorthead redhorse, *Moxostoma macrolepidotum*), 1800 (golden redhorse, *M. erythrurum*), and 6000 (minnow, *Notropis* sp.). Continuous exposures of *P. promelas* to mean trifluralin concentrations of 5.1, 1.9, and 1.5 $\mu g \cdot L^{-1}$ for 425 d resulted in eviscerated carcass BCFs of 961, 1333, and 889, respectively (Macek *et al.* 1976).

Elimination of trifluralin from Atlantic salmon (*Salmo salar*) fry following an 11-h exposure to 0.5 $mg \cdot L^{-1}$ followed first-order kinetics with a rate constant of 0.017 d^{-1} resulting in a tissue half-life of 40.5 d (Wells and Cowan 1982).

Microcosms simulating a northern prairie wetland were exposed to 4 $\mu g \cdot L^{-1}$ ^{14}C-trifluralin in a sediment water mixture by Huckins, Petty, and England (1986). Although trifluralin in the water column decreased below the minimum detectable limit of 0.5 $\mu g \cdot L^{-1}$ after 7 d, residues in the form of ^{14}C degradation products were quantifiable at the end of the 6-week experimental period. *Daphnia* accumulated the highest concentration of ^{14}C-trifluralin/trifluralin metabolites (566 $ng \cdot g^{-1}$) of all the organisms in the microcosms. Midge larvae *(Chironomus riparius)* macrophytes and algae contained trifluralin/trifluralin metabolites in the range of 40 to 260 $ng \cdot g^{-1}$ (Huckins, Petty, and England 1986). Calculation of BCFs from these data was not possible.

Information related to bioconcentration of trifluralin in terrestrial organisms was not found.

Acute Toxicity to Aquatic Organisms

A summary of the aquatic acute toxicity data for trifluralin is presented in Table E-1. The tests reported in this table were conducted with trifluralin dissolved in water with or without a solvent carrier. These tests did not consider the effect of suspended solids on trifluralin toxicity as the test procedures usually used filtered natural waters or dechlorinated tap water.

The vertebrate acute toxicity data base for trifluralin consists of fifty 24-h LC_{50} values, two 48-h LC_{50} values, and fifty-seven 96-h LC_{50} values. Of the fifty-seven 96-h tests, two used the larval stage of an amphibian (tadpole) and the remainder were conducted with seven species of freshwater fish. Twenty-five tests used various life stages of the rainbow trout *(Salmo gairdneri)*. Hashimoto and Nishiuchi (1982) reported a 48-h LC_{50} of 1.0 $mg \cdot L^{-1}$ for the carp, *Cyprinus carpio*, and a 48-h LC_{50} of 14 $mg \cdot L^{-1}$ for the tadpole of the frog *Bufo bufo japonicus*. These tests were not presented in Table E-1 because details of the tests were not provided.

Acute toxicity studies by Mayer and Ellersieck (1986) using the rainbow trout (*Salmo gairdneri*) have shown a decrease in the trifluralin LC_{50} (from 560 to 100 $\mu g \cdot L^{-1}$) with an increase in temperature of the bathing water, which is indicative of an increased metabolic rate.

Toxicity tests using trifluralin adsorbed onto soil, instead of dissolved in the water, required as much as 227 times the amount of trifluralin to produce 50% mortality among bluegill (*Lepomis macrochirus)* (Parka and Worth 1965). They concluded that the possibility of acutely toxic quantities of trifluralin washing into an aquatic environment from an adjacent treated field is remote. Using their LC_{50} value for trifluralin adsorbed to soil, Parka and Worth (1965) calculated that over 13 million kg of soil treated with 0.56 $kg \cdot ha^{-1}$ would have to wash into a 0.4-ha pond with an average depth of 0.9 m to produce 50% mortality among the fish population.

Invertebrate acute toxicity data for trifluralin consist of six 24-h LC_{50} values, twenty 48-h LC_{50} values, and ten 96-h LC_{50} values from tests using 14 species of freshwater invertebrates and 1 species of estuarine mollusc (Table E-1).

Information related to the acute toxicity of trifluralin to aquatic plants is scarce. Significant (96%) decreases in the growth of populations of a single cell green alga, *Chlamydomonas eugametos*, as measured by cell counts, was caused by 335.5 $\mu g \cdot L^{-1}$ trifluralin. Significant changes in growth were not observed at 33.55 $\mu g \cdot L^{-1}$ (Hess 1980). A 50% decrease in the optical density of the green flagellated alga *Dunaliella bioculata* was also produced by a trifluralin concentration of 335.5 $\mu g \cdot L^{-1}$ (Felix, Chollet, and Harr 1988). Data related to the acute toxicity of trifluralin to aquatic vascular plants were not found.

Chronic Toxicity and Sublethal Reactions

Vertebrate chronic toxicity and sublethal reaction data include long-term (12–570 d) exposures using the freshwater fathead minnow (*Pimephales promelas)* and the estuarine sheepshead minnow (*Cyprinodon variegatus).* In addition, long-term (12-month) sublethal reactions were also observed in Atlantic salmon (*Salmo salar*) initially exposed to sublethal levels of trifluralin for less than 12 h. A summary of studies dealing with long-term exposure is presented in Table F-1.

Continuous exposures of fathead minnows for 125–158 d to a mean concentration of 8.2 $\mu g \cdot L^{-1}$ produced 100% mortality among the 40 test fish. Over half of the fish exposed to mean concentrations of 5.1 $\mu g \cdot L^{-1}$ died during the 163- to 263-d portion of the 425-d test period. Surviving fish spawned 100 d later than the control fish and fish exposed to 1.9 $\mu g \cdot L^{-1}$. Based on survival, the estimated maximum acceptable toxicant concentration (MATC) for this species is between 1.95 and 5.1 $\mu g \cdot L^{-1}$ (Macek *et al.* 1976).

Long-term (28–570 d) exposure to low recurrent trifluralin concentrations (1–6 $\mu g \cdot L^{-1}$) caused abnormalities in vertebral development and other histopathological effects in sheepshead minnows. Vertebral dysplasia occurred in sheepshead minnows exposed to 5.5 $\mu g \cdot L^{-1}$ during the first 28 d of life from the zygote stage (Couch *et al.* 1979). An investigation into the possible role of the pituitary gland in the trifluralin-induced vertebral lesions discovered histopathological changes in, and enlargement of, the pituitary gland in 11 out of 20 sheepshead minnows exposed to 1–5 $\mu g \cdot L^{-1}$ for 30 d to 19 months. These changes, however, could not be definitively linked to the observed vertebral lesions (Couch 1984).

In the laboratory, Atlantic salmon were exposed to 0.5 $mg \cdot L^{-1}$ trifluralin for 11 h and observed for the following 12 months (Wells and Cowan 1982). The fish exhibited a rapid uptake of trifluralin, and concentrations of trifluralin in the fish of approximately 100 $mg \cdot kg^{-1}$ whole weight were retained for several days. Of the 100 fish exposed to the 11-h dose, 9 died soon after exposure ceased and the survivors appeared to be more susceptible to fungal infection. The results of measurements taken from X-ray plates showed vertebral deformation when the trifluralin concentration in the fish was at a maximum (approximately 100 $mg \cdot kg^{-1}$). There was no apparent increase in the degree of deformation during the first month after treatment. Subsequent measures of fish growth over the following 11 months indicated that trifluralin caused a contraction of the vertebral column resulting in the loss of the normal fusiform shape of the fish and the development of a more truncated shape. These same effects were observed in a natural population of brown trout (*Salmo trutta)* as the result of an accidental spill of a trifluralin-containing herbicide (Wells and Cowan 1982). Trifluralin concentrations in the water as a result of the spill were not reported.

Trifluralin caused a 63% reduction *in vitro* in the sodium uptake by perfused carp *(Cyprinus carpio)* gills. The exact concentration of trifluralin causing this decrease was unknown, but was assumed to be less than the trifluralin solubility in carp Ringer solution (500 $\mu g \cdot L^{-1}$) (McBride and Richards 1975).

Invertebrate chronic toxicity is represented by trifluralin exposures of 64 d for the freshwater cladoceran *Daphnia magna* and 80-d exposures of the burrowing aquatic oligochaete *Limnodrilus hoffmeisteri* to trifluralin-contaminated sediment. These studies are summarized in Table F-1.

Continuous exposure of *D. magna* to 7.2 $\mu g \cdot L^{-1}$ over three generations reduced survival with survival decreasing with each generation. None of the third generation animals survived. Production of young per adult exposed to 14.0 $\mu g \cdot L^{-1}$ during the first two generations was also reduced (52%–69%). Based on survival, the estimated MATC for *D. magna* continuously exposed through three generations is between 2.4 and 7.2 $\mu g \cdot L^{-1}$ (Macek *et al.* 1976).

A study to examine the impact of tubificid worms on pollutant transport in sediment demonstrated that a trifluralin sediment concentration of 1.2 $mg \cdot kg^{-1}$ did not affect the survival or normal functioning of these burrowing worms (Karickhoff and Morris 1985).

A variety of chronic toxicity data is available for marine and estuarine invertebrates. For example, an MATC for the zoeal stage of the dungeness crab (*Cancer magister*) was determined to be between 26 and 220 $\mu g \cdot L^{-1}$ for an 80-d exposure (Caldwell *et al.* 1979). Liu and Lee (1975) concluded that trifluralin may be lethal to adult bay mussels (*Mytilus edulis*) at 240 $\mu g \cdot L^{-1}$ after 4-d exposures and inhibitory to the larval stage of this mussel at 96 $\mu g \cdot L^{-1}$ if exposure exceeds 10 d.

Community Studies

The toxic effect of trifluralin on various types of aquatic communities has been investigated using

microcosms. Single doses producing a final concentration of 10 000 $\mu g \cdot L^{-1}$ trifluralin had no effect on algal communities in artificial outdoor recirculating streams during a 3-week period (Kosinski 1984; Kosinski and Merkle 1984). A single dose of trifluralin resulting in an initial concentration of 1000 $\mu g \cdot L^{-1}$ in a wetland microcosm did not adversely affect phytoplankton populations, gross primary productivity, or macrophytes *Lemna* sp. *Ceratophyllum* sp., and *Elodea* sp. As well, respiratory electron transport system activity, metabolism of organic carbon, oxygen consumption, and phosphate activity were not affected by 1000 $\mu g \cdot L^{-1}$ trifluralin over the 30-d observation period (Johnson 1986).

A total of 5 mg trifluralin, containing ^{14}C-labelled carbon, was injected to a depth of 1 cm in the soil of a terrestrial microcosm (Cole and Metcalf 1980). This produced a total residue of 0.224 $mg \cdot kg^{-1}$ in a vole (*Michrotus ochrogaster)* exposed to the contaminated environment for 5 d. Of this residue, 27% was the parent trifluralin. Additional residues in terrestrial animals were 4.29 $mg \cdot kg^{-1}$ for earthworms (*Lumbricus terrestris)* and 0.472 $mg \cdot kg^{-1}$ for slugs *(Limex maximus)*. After 20 d, the terrestrial microcosm was flooded with water and maintained as an aquatic microcosm for 7 d with snails (*Physa* sp.) and mosquitofish. After 7 d, the snails contained a residue of 0.571 $mg \cdot kg^{-1}$, of which 0.171 $mg \cdot kg^{-1}$ was the parent trifluralin. Total residues in the fish were 0.059 $mg \cdot kg^{-1}$, of which 0.007 $mg \cdot kg^{-1}$ was the parent trifluralin. The fish were rapidly killed after being introduced to the microcosm (within 4 h), but the authors note that the lethal substance was not the parent trifluralin but some toxic metabolite; the water contained a ^{14}C residue of 9.13 $\mu g \cdot L^{-1}$, but none of this was the parent compound.

Microcosms containing uncontaminated soil, snails (*Helosoma* sp.), algae (*Oedogonium cardiacum*), and mosquitofish (*Gambusia affinis*) received continuous inputs of ^{14}C-trifluralin for 30 d. The highest input of trifluralin caused water concentrations ranging from 9.3 $\mu g \cdot L^{-1}$ at day 2 to 29.8 $\mu g \cdot L^{-1}$ at day 15 over the 30-d test period. During this time, the inhibition of growth was observed visually in comparison to the control microcosms. Both the fish and snails reproduced during the test period. The fish offspring were observed to behave abnormally and had an unusual curvature of the back and darkening of the tail region. Adult fish also exhibited abnormal behaviour and spinal curvature. This concentration range was not acutely toxic and all fish survived for 67 d after trifluralin inputs were terminated (Yockim, Isensee, and Walker 1980).

Microcosms consisting of naturally coadapted communities of phytoplankton, bacteria, zooplankton, and small benthic invertebrates (designed for screening the ecological impacts of pesticides on community functions) received a single dose of trifluralin, which produced an initial concentration of 200 $\mu g \cdot L^{-1}$ (Sheehan, Axler, and Newhook 1986). For one set of tests over a 14-d period, the electron transport system (ETS) potential activity (mg $O_2 \cdot L^{-1} \cdot h^{-1}$) of the trifluralin-treated microcosms remained within the range of ETS activity defined by the control microcosms. Primary productivity was reduced below the control range when measured on days 4, 7, and 14, but recovered to within the control range at 28 d. Maximum deviation from the control occurred at 7 d.

Guideline

The derivation of the guideline value for freshwater aquatic life was initiated with the lowest or most sensitive MATC from the literature. The lower limit of the MATC for the 425-d trifluralin exposure for fathead minnows is 1.95 $\mu g \cdot L^{-1}$ (Macek *et al.* 1976). Thus, 1.95 $\mu g \cdot L^{-1}$ or 2 $\mu g \cdot L^{-1}$ is used to define the lowest-observed-effect concentration (LOEC).

Given the wide range of half-lives reported for trifluralin in the environment, some of which indicate that the compound is persistent, plus the bioaccumulation potential of this compound, it is appropriate that a safety factor of 0.1 level of magnitude be used to derive a guideline for the protection of freshwater organisms. Use of the application factor with the LOEC value of 2 $\mu g \cdot L^{-1}$ produces a guideline value of 0.2 $\mu g \cdot L^{-1}$.

Agricultural Water Supply

Livestock Watering

Acute Toxicity

Trifluralin exhibits low acute oral toxicity to mammals and birds with LD_{50} values for mice above 5 $g \cdot L^{-1}$ (U.S. EPA 1984, 1987b). Fertilized mallard (*Anas platyrhynchos)* eggs were used in embryo acute toxicity tests for trifluralin. Immersion of eggs for 30 s

at room temperature into various aqueous emulsions of trifluralin resulted in an LC_{50} equivalent of 1.8 kg·ha^{-1}. Doses equal to or greater than the LC_{50} reduced embryo growth and produced abnormalities in morphology at 18 d (Hoffman and Albers 1984). Information concerning acute trifluralin toxicity to livestock was not found.

Subacute and Chronic Toxicity

Long-term trifluralin ingestion studies have generally been conducted with laboratory studies using rats, mice, and dogs. A 90-d feeding study using female rats continuously fed trifluralin at 50 and 100 mg·kg^{-1}·d^{-1} produced a no-observed-adverse-effect level (NOAEL) of 25 mg·kg^{-1}·d^{-1}, based on increased liver weights of the progeny (U.S. EPA 1987b). Another NOAEL of 100 mg·kg^{-1}·d^{-1} resulted from a 729-d trifluralin ingestion study using rat growth rate, mortality, and food consumption as effect criteria. Rats consuming 1000 mg·kg^{-1}·d^{-1}, the next highest dose, exhibited reduced food consumption and displayed a slight proliferation of bile duct tissue. Other histopathological or hematological effects were not observed (U.S. EPA 1987b). A 2-year trifluralin ingestion study in male and female rats produced a NOAEL of 30–37 mg·kg^{-1}·d^{-1}. Body weight, food consumption, hemoglobin, and red blood cell counts were decreased, and blood urea nitrogen, liver weight, and testes weights were increased at 128-mg·kg^{-1}·d^{-1} (male) and 154-mg·kg^{-1}·d^{-1} (female) dose rates. Kidney and heart weights were also decreased in females (U.S. EPA 1987b).

Based on the hematology, body, kidney, and spleen weights in both sexes and uterine weights in females, mice ingesting doses of trifluralin for 2 years exhibited a NOAEL of 40 mg·kg^{-1}·d^{-1}. No increase in vomiting or liver-to-body weight ratios was observed in dogs fed 10 mg·kg^{-1}·d^{-1} during a 3-year continuous trifluralin ingestion study (U.S. EPA 1987b).

A lowest-observed-adverse-effect level (LOAEL) of 2.5 mg·kg^{-1}·d^{-1} was derived from a 90-d study of male rats in which increases in α-1, α-2, and β-globulins were monitored in the blood. Lower levels of trifluralin ingestion were not tested. Other effects observed at levels equal to or greater than 160 mg·kg^{-1}·d^{-1} were increased levels of aspartate transaminase and urinary calcium, inorganic phosphorus, and magnesium (U.S. EPA 1987b).

A conference paper presented by the manufacturer of trifluralin stated that rats fed diets containing trifluralin levels as high as 2000 mg·kg^{-1} feed (approximately 100 mg·kg^{-1}·d^{-1}) for 2 years did not exhibit changes in blood hematocrit, hemoglobin, total blood cell numbers, organ weight ratios, or gross and microscopic histology. The same diet through three generations of rats also failed to produce treatment-related effects (Worth and Anderson 1965). This paper, however, did not give specific documentation of the studies. Histopathological changes in mouse kidney were observed after ingestion of trifluralin at 14, 140, and 1400 mg·kg^{-1}·d^{-1} for 140 d. Degeneration of proximal and distal tubule cells was observed at all dosages and the amount of degeneration was dose-related (Akay 1986).

Uptake, Metabolism, and Elimination

Trifluralin is not readily absorbed from the gastrointestinal (GI) tract, and the fraction that is absorbed is completely metabolized. Low GI tract absorption of a single oral dose of 100 mg·kg^{-1} body weight was indicated by 11%–14% excretion in the bile after 24 h (Emmerson and Anderson 1966). Although sufficient data are not available to completely characterize mammalian or avian trifluralin metabolism, four metabolites have been identified in rats. These metabolites were the result of the removal of the N-propyl groups and/or reduction of the nitro groups to amine groups (Emmerson and Anderson 1966). This is in agreement with the results of *in vitro* studies using rat hepatic microsomes (Nelson *et al.* 1977) and ^{14}C-trifluralin administration to a cow and two goats (Golab *et al.* 1969).

Elimination of oral doses of trifluralin in rats is mainly via the feces. Approximately 78% of an oral dose of 100 mg·kg^{-1} was eliminated from rats via this route while the remainder was eliminated in the urine. Virtually all of the dose was excreted in 3 d (Emmerson and Anderson 1966).

Trifluralin with a ^{14}C-trifluoromethyl group was administered to a lactating cow at a dietary concentration of 1 mg·kg^{-1} for 39 d followed by 1000 mg·kg^{-1} for 13 d. In contrast to studies using rats, only trace quantities of trifluralin and several trifluralin metabolites were found in the feces, but 99% of the ingested radioactivity was recovered in the urine within 6 d. A maximum concentration of 6.5 mg·kg^{-1} trifluralin in feces was found 6 d after initiation of the 1000–mg·kg^{-1} dose. Metabolites were approximately

21 $mg \cdot L^{-1}$ feces at the same time (Golab *et al.* 1969). Any observed toxic responses were not discussed.

In a 26-d experiment, two lactating goats were fed unlabelled trifluralin at 1 $mg \cdot kg^{-1}$ body weight for 11 d and received ^{14}C-trifluralin on day 12 followed by unlabelled trifluralin for the remaining 14 d. The unlabelled trifluralin was not ingested in sufficient quantities for the identification of trifluralin metabolites. The ^{14}C-trifluralin administration revealed that 17.8% and 81.2% of the trifluralin and metabolites, however, were eliminated in the urine and feces, respectively. The ^{14}C-trifluralin and metabolites appeared in the urine for 3 d and in the feces for 6 d after ingestion. There was a 99% recovery of the labelled material (Golab *et al.* 1969). Any observed toxic responses were not discussed.

Carcinogenicity, Mutagenicity, and Teratogenicity

Trifluralin may be classified as a possible human carcinogen as it shows limited evidence of carcinogenicity in animals. Present evidence for human carcinogenicity, however, is lacking. Dose-related increases in hepatocellular carcinomas and alveolar adenomas were observed in female mice exposed to 33 or 62 $mg \cdot kg^{-1} \cdot d^{-1}$ trifluralin in the diet for 1.5 years. The trifluralin technical product used in this study, however, contained 84–88 $mg \cdot kg^{-1}$ dipropyl-nitrosamine (DPNA), a known carcinogen in rats and mutagen in bacterial and cell culture systems (U.S. EPA 1984). The issue of DPNA contamination of trifluralin resulted in the proposed cancellation of registration of all products containing trifluralin by the U.S. Environmental Protection Agency if DPNA could not be reduced to a level at or below 1 $mg \cdot kg^{-1}$ (U.S. EPA 1979). Subsequently, the manufacturer lowered the level of DPNA in trifluralin and conducted long-term ingestion studies with rats and mice. The manufacturer's 2-year dietary carcinogenicity assay with mice, using trifluralin containing <0.01 $mg \cdot kg^{-1}$ DPNA, reported no treatment-related increases in benign or malignant neoplasms (U.S. EPA 1984, 1987a). Use of the same low-level DPNA trifluralin in a 2-year rat ingestion study showed increases in kidney, urinary bladder, and thyroid tumors in male rats receiving 30, 128, or 272 $mg \cdot kg^{-1} \cdot d^{-1}$. Based on these studies and a reevaluation of the risk posed to individuals working with trifluralin, the U.S. EPA decided that the risks associated with the development of cancer as a result of trifluralin exposure were not excessively high. They concluded that the benefits of trifluralin use outweighed the identifiable risks involved with its registration if the total DPNA level in the technical product could be maintained at or below 0.5 $mg \cdot kg^{-1}$ (U.S. EPA 1982). The U.S. EPA Carcinogen Assessment Group estimated a carcinogenic potency factor of 0.00766 per $mg \cdot kg^{-1} \cdot d^{-1}$ and an estimated life-time cancer risk of 10^{-6} for consumption of water containing 5 $\mu g \cdot L^{-1}$ trifluralin (U.S. EPA 1987b). This corresponds to one additional case of cancer in a population of one million people.

Genotoxicity testing of trifluralin in several *in vitro* and *in vivo* systems was negative with and without metabolic activation. These systems included *Salmonella typhimurium* and *Escherichia coli* reverse mutation assays, mouse lymphoma cells, and Chinese hamster ovary sister chromatid exchange (Andersen, Leighty, and Takahashi 1972; U.S. EPA 1987b).

Trifluralin was demonstrated to increase the incidence of chromosome nondisjunction *in vivo* in the fruit fly (*Drosophila melanogaster*) (Murnik 1978; Bryant and Murnik 1979). In addition to nondisjunction, trifluralin has also been reported to produce deletion of the paternal X or Y chromosome in the progeny of male fruit flies fed trifluralin as larvae (Foureman 1981a, 1981b). Equivocal results for the sex-linked recessive lethal mutagenic assay were also produced with fruit flies (Yoon *et al.* 1985). Trifluralin apparently caused nondisjunction by interfering with the cytokinetic mechanism for separating the replicated chromosomes during cell division (Sentein 1977; Merezhinskii and Sharmankin 1986). This is similar to the cytokinetic mode of action in plants.

Increased chromosomal abnormalities in bone marrow cells, evidence of gametic mutation in the spermatocyte test, positive results in the dominant lethal assay, and alterations of F_1 embryonic chromosomes were cited by Nehez *et al.* (1980, 1981) to be the result of treatments of mice with the herbicide OLITREF®, which contains 26% trifluralin. The extent to which DPNA influenced the results reported by Nehez *et al.* (1980, 1981) is unknown. DPNA, however, is a common contaminant of commercial products containing trifluralin (U.S. EPA 1984). It is thought that DPNA is not responsible for the fruit fly chromosomal nondisjunction discussed above (U.S. EPA 1984).

Teratological studies using rabbits conducted by the manufacturer of trifluralin reported a NOAEL of 225 $mg \cdot kg^{-1} \cdot d^{-1}$ for maternal and reproductive effects. Higher doses of 500 and 800 $mg \cdot kg^{-1} \cdot d^{-1}$ caused

anorexia and cachexia in the females and aborted litters at dosages of 225 $mg \cdot kg^{-1} \cdot d^{-1}$ (U.S. EPA 1987b). Despite these studies, the manufacturer identified 1000-$mg \cdot kg^{-1} \cdot d^{-1}$ trifluralin ingestion as a reproductive "safe level," citing no effects on litter size and weight in 32 pregnant rabbits receiving this dosage (U.S. EPA 1987b). The confidential nature of these reports did not permit further data analysis.

Exposure of female mice to trifluralin on each of gestational days 6–15 resulted in a significant (19%) increase in skeletal abnormalities in their progeny at 62 d post-partum. Doses of 1.0 $mg \cdot kg^{-1}$ body weight in corn oil were administered by gavage (Beck 1977, 1981). The possible influence of DPNA in these studies was not discussed.

Guideline

An LOAEL of 2.5 $mg \cdot kg^{-1} \cdot d^{-1}$ was generated from a 90-d study with laboratory rats using changes in blood globulin levels as an effect criterion (U.S. EPA 1987b). If this value is used with a safety factor of 0.01, the assumed safe level would be 0.025 $mg \cdot kg^{-1} \cdot d^{-1}$. Using the weight and water consumption of a dairy cow (500 kg and 160 $L \cdot d^{-1}$), the concentration of 0.025 $mg \cdot kg^{-1} \cdot d^{-1}$ translates to a water concentration of 78 $\mu g \cdot L^{-1}$. This is just under one half the drinking water guideline for trifluralin of 170 $\mu g \cdot L^{-1}$ established by the World Health Organization (WHO 1987). The value of 78 $\mu g \cdot L^{-1}$ was derived from a rate of trifluralin ingestion less than the lowest known NOAEL in the scientific literature. This, plus the generally limited absorption of trifluralin by the GI tract and its rapid metabolism, could make the value of 78 $\mu g \cdot L^{-1}$ an appropriate interim guideline value. Additional data related to the long-term ingestion of trifluralin by livestock via drinking water will be required prior to the development of a guideline value. In the interim, the procedure recommended by CCREM (1987) of adopting the drinking water guideline for livestock watering in the absence of sufficient information is followed. In the case of trifluralin, an interim drinking water quality guideline for trifluralin of 45 $\mu g \cdot L^{-1}$ has been proposed, and this is also recommended as a livestock watering guideline.

Irrigation

Toxicity to Nontarget Plant Species

At the whole plant level, a large number of studies have described the toxicity of trifluralin to nontarget plants using a wide variety of criteria in addition to lethality. Some of these studies are presented in Table G-1. The data in this table demonstrate that a wide variety of crops, including those considered tolerant, are susceptible to the toxic effects of trifluralin given the proper dosage and conditions. Under the routine preplant incorportion conditions, trifluralin causes its greatest phytotoxic effect on the meristematic tissue at the region of the coleoptile node (Billett and Ashford 1978).

Field cultivation, greenhouse, and growth chamber studies have demonstrated adverse, sublethal reactions of seedling plants to applications as low as 0.56 $kg \cdot ha^{-1}$ and water concentrations as low as 90 $\mu g \cdot L^{-1}$. The only generalization that can be drawn from the phytotoxicity data is that specific soil conditions and plant species are major factors in determining the potential for plant injury. The action of trifluralin on the root system may also induce stress on the plant related to its ability to obtain sufficient nutrients. This type of reaction was demonstrated by Udoh and Nelson (1986) for site-specific iron deficiency in soybeans. The bioavailability of trifluralin in soil is mainly dependent on the amount of soil organic matter. Increasing organic matter causes a decrease of trifluralin efficacy at constant levels of soil moisture. Clay content and temperature have no effect. Variation in moisture levels also plays a small role in trifluralin bioavailability (Moyer 1979). At low concentrations (3.4 $\mu g \cdot L^{-1}$), trifluralin can be metabolized by some plants resulting in N-didealkylated products and/or a para-carboxylic acid derivative (Camper, Ahmed, and Figliola 1989).

Guideline

Much of the terrestrial phytotoxicity data was generated on the basis of the weight of trifluralin applied to a surface area of soil. It is difficult to extrapolate these units to the concentration of trifluralin in water that would be detrimental to irrigated crops.

Phytotoxicity studies with trifluralin in water used to irrigate a soil that supported seeds and/or seedlings showed that relatively small quantities of trifluralin, as low as 90 $\mu g \cdot L^{-1}$, could cause detrimental responses in the root growth of some plant species (Barrentine and Warren 1971). This study, however, used acetone as a carrier for trifluralin because of its low solubility in water. Although implied, the presence of acetone controls was not specifically identified. In addition, the matrix used for seed germination and seedling growth was silica sand, which allowed minimal adsorption of the trifluralin and maximum exposure of the plant

tissue. These conditions are unlike any that would be encountered in the field and represent extremes of toxicity. Thus, these values cannot be used to derive a guideline value for the protection of irrigated crops. A solvent carrier for trifluralin was not employed by Harvey (1973), who used trifluralin-amended Hoagland's nutrient solution to grow soybeans from seed in silica sand. This greenhouse study demonstrated that a trifluralin concentration of 3.35 $mg \cdot L^{-1}$ caused a 40% decrease in the dry weight of the seedling plants after 28 d. Unfortunately, this was the only concentration of trifluralin used and a NOEL was not defined.

Given trifluralin's low water solubility, high volatility, and sediment/water distribution coefficients, it is doubtful that sufficient quantities of trifluralin could be maintained in irrigation water to be harmful to plants. There is insufficient information, however, to support a specific guideline or interim guideline value for trifluralin in irrigation water.

Recreational Water Quality and Aesthetics

Organoleptic Effects

Although volatilization is a major transport pathway for dispersion of trifluralin in the environment, reports dealing with trifluralin-caused taste and odour in water were not found. Trifluralin is also known to be rapidly accumulated by fish, but reports dealing with trifluralin-caused tainting of fish flesh were not found.

Guideline

At present, there is no evidence to indicate that recreational water use would be adversely affected by trifluralin residues when this herbicide is used according to label instructions. In addition, water containing trifluralin residues at concentrations that could potentially affect recreational water uses would already be severely impaired for other water uses (i.e., water for the preservation of aquatic life). Thus, a water quality guideline has not been determined for recreational waters and aesthetics.

Industrial Water Supplies

Guideline

To date, there is no indication that trifluralin poses or has the potential to pose a threat to the quality of water used for industry when used according to registered use patterns. Although of potential concern if found in water supplies, a water quality guideline for trifluralin in industrial water supplies has not been determined.

SUMMARY

After an evaluation of the published information on the herbicide trifluralin, the Canadian water quality guidelines were derived (Table 2). The background information on trifluralin in terms of uses and production, occurrence in the aquatic environment, and persistence and degradation was reviewed. The rationale employed for the development of the recommended guidelines was summarized.

Table 2. Recommended Water Quality Guidelines for Trifluralin

Uses	Guidelines
Raw water for drinking water supply	45 $\mu g \cdot L^{-1}$ (IMAC)*
Freshwater aquatic life	0.2 $\mu g \cdot L^{-1}$
Agricultural water supply	
Livestock watering	45 $\mu g \cdot L^{-1}$ (Interim)
Irrigation	No recommended guideline
Recreational water quality and aesthetics	No recommended guideline
Industrial water supplies	No recommended guideline

*Existing drinking water guideline (Health and Welfare Canada 1987).

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REFERENCES

Agriculture Canada. 1989. Regulatory information on pesticide products. RIPP Database (CCINFODISK). Produced by Agriculture Canada and distributed by the Canadian Centre for Occupational Health and Safety. (CD-ROM.)

Akay, M.T. 1986. Histological investigation of kidney degeneration caused by trifluralin in mice. Doga Biyol. Ser. 10(1): 1–7 (abstract).

Alabaster, J.S. 1969. Survival of fish in 164 herbicides, insecticides, fungicides, wetting agents and miscellaneous substances. Int. Pest Control 11(2): 29–35.

Andersen, K.J., E.G. Leighty, and M.T. Takahashi. 1972. Evaluation of herbicides for possible mutagenic properties. J. Agric. Food Chem. 20(3): 649–656.

Anderson, W.P., A.B. Richards, and J.W. Whitworth. 1967. Trifluralin effects on cotton seedlings. Weeds 15: 224–227.

Ashley, R.A. 1973a. Evaluation of herbicides and herbicide combinations for weed control in seeded cucumbers and summer squash. Proc. North. Weed Sci. Soc. 27: 226–231.

Ashley, R.A. 1973b. Two-year study of weed control in snap beans. Proc. North. Weed Sci. Soc. 27: 178–183.

Ashton, F.M., and A.S. Crafts. 1973. Mode of action of herbicides. Toronto: John Wiley & Sons.

Ashton, F.M., D. Penner, and S. Hoffman. 1968. Effect of several herbicides on proteolytic activity of squash seedlings. Weed Sci. 16: 169–171.

Ashton, F.M., O.T. de Villiers, R.K. Glenn, and W.B. Duke. 1977. Localization of metabolic sites of action of herbicides. Pestic. Biochem. 7: 122–141.

Axe, J.A., A.C. Mathers, and A.F. Weise. 1969. Disappearance of atrazine, propazine and trifluralin from soil and water. Proc. South. Weed Sci. Soc. 22: 367 (abstract).

Banks, P.A., and P.W. Santlemann. 1978. Influence of subsurface layered herbicides on horsenettle and various crops. Agron. J. 70: 5–8.

Bardsley, C.E., K.E. Savage, and V.O. Childers. 1967. Trifluralin behaviour in soil. I. Toxicity and persistence as related to organic matter. Agron. J. 59: 159–160.

Barrentine, W.L., and G.F. Warren. 1971. Differential phyto-toxicity of trifluralin and nitralin. Weed Sci. 19: 31–37.

Basler, E., and P.W. Santlemann. 1975. Effects of trifluralin on the absorption and translocation of phosphorate and rubidium by soybean seedlings. Proc. South. Weed Sci. Soc. 28: 316–319.

Bayer, D.E., C.L. Foy, T.E. Mallory, and E.G. Cutter. 1967. Morphological and histological effects of trifluralin on root development. Am. J. Bot. 54(8): 945–952.

Beck, S.L. 1977. Postnatal detection of prenatal exposure to herbicides in mice using normally occurring variations in skeletal development. Teratology 15: 15A (abstract).

Beck, S.L. 1981. Assessment of adult skeletons to detect prenatal exposure to 2,4,5-T or trifluralin in mice. Teratology. 23: 33–55.

Behran, S., M. Maftoun, B. Sheibany, and S.M. Hojjati. 1979. Effect of fertilizer-N and herbicides on the growth and N content of soybeans and cowpeas. Agron. J. 71: 533–538.

Beste, C.E. 1974. Weed control in transplanted tomatoes. Proc. North. Weed Sci. Soc. 28: 265–269.

Beste, C.E. 1975. New herbicides for lima beans. Proc. North. Weed Sci. Soc. 29: 194–196.

Billett, D., and R. Ashford. 1978. Differences in the phytotoxic response of wild oats (*Avena fatua*) to triallate and trifluralin. Weed Sci. 26(3): 273–276.

Bollich, P.R., E.P. Dunigan, L.M. Kitchen, and V. Taylor. 1988. The influence of trifluralin and pendimethalin on nodulation, $N_2(C_2H_2)$ fixation, and seed yield of field-grown soybeans (*Glycine max*). Weed Sci. 36: 15-19.

Bradley, M.R., and G. Hargreaves. 1977. Weed control in drilled outdoor lettuce. Exp. Hortic. 29: 35–44.

Brown, J.F., and H.D. Swingle. 1977. Herbicide evaluation in vegetable crops. Proc. South. Weed Sci. Soc. 30: 168–175.

Bryant, M.L., and M.R. Murnik. 1979. Mutagenicity of the herbicide trifluralin in *Drosophila melanogaster*. Genetics 91 (Suppl.): 15.

Bucholtz, D.L., and T.L. Lavy. 1978. Pesticide interactions in oats (*Avena sativa* L. 'Neal'). J. Agric. Food Chem. 26(3): 520–524.

Bucholtz, D.L., and T.L. Lavy. 1979. Alachlor and trifluralin effects on nutrient uptake in oats and soybeans. Agron. J. 71: 24–26.

Burnside, O.C. 1968. Control of wild cane in soybeans. Weed Sci. 16: 18–22.

Burnside, O.C. 1974. Trifluralin dissipation in soil following repeated annual applications. Weed Sci. 22(4): 374–377.

Bush, J.D., J.R. Abernathy, and J.R. Gipson. 1982. Adsorption and mobility of trifluralin and pendimethalin in west Texas soils. Proc. South. Weed Sci. Soc. 35: 340.

Caldwell, R.S., D.V. Buchanan, D.A. Armstrong, M.H. Malon, and R.E. Millemann. 1979. Toxicity of the herbicides 2,4-D, DEF, propanil and trifluralin to the dungeness crab, *Cancer magister*. Arch. Environ. Contam. Toxicol. 8: 383–396.

Camanzo, J., C.P. Rice, D.J. Jude, and R. Rossman. 1987. Organic priority pollutants in nearshore fish from 14 Lake Michigan tributaries and embayments, 1983. J. Great Lakes Res. 13(3): 296–309.

Camper, N.D., F.A. Ahmed, and S. Figliola. 1989. Growth effects, uptake and metabolism of trifluralin in tissue cultures. J. Environ. Sci. Health 24: 291–306.

Camper, N.D., K. Stralka, and H.D. Skipper. 1980. Aerobic and anaerobic degradation of profluralin and trifluralin. J. Environ. Sci. Health B15: 457–473.

Cassidy, J.C. 1972. Herbicide evaluation in *Brassica* crops. Proc. Br. Weed Control Conf. 11: 958–964.

CCREM (Canadian Council of Resource and Environment Ministers). 1987. Canadian water quality guidelines. Prepared by the Task Force on Water Quality Guidelines of the Canadian Council of Resource and Environment Ministers.

Clark, J.R., D. DeVault, and R.J. Bowden. 1984. Contaminant analysis of fillets from Great Lakes coho salmon, 1980. J. Great Lakes Res. 10: 38–47.

Cole, L.K., and R.L. Metcalf. 1980. Environmental destinies of insecticides, herbicides, and fungicides in the plants, animals, soil, air, and water of homologous microcosms. In: Microcosms in ecological research, ed. J.P. Giesy, Jr. Technical Information Center, U.S. Department of Energy.

Conn, J.S., and C.W. Knight. 1984. An evaluation of herbicides for broadleaf control in rapeseed. Efficacy, phytotoxicity and soil persistence studies. Bulletin 62. Agricultural Experimental Station, School of Agriculture and Land Resources Management, University of Alaska-Fairbanks.

Cope, O.B. 1965. Some responses of freshwater fish to herbicides. Proc. South. Weed Conf. 18: 439–445.

Cope, O.B. 1966. Contamination of the freshwater ecosystem by pesticides. J. Appl. Ecol. 3 (Suppl.): 33–44.

Couch, J.A. 1984. Histopathology and enlargement of the pituitary in a teleost exposed to the herbicide trifluralin. J. Fish Dis. 7: 157–163.

Couch, J.A., L.A. Courtney, and S.S. Foss. 1981. Laboratory evaluation of marine fishes and carcinogen assays subjects. In: Phyletic approaches to cancer, ed. C.J. Dawe *et al.* pp. 125-139. Tokyo: Japan Science Society Press.

Couch, J.A., J.T. Winstead, D.J. Hansen, and L.R. Goodman. 1979. Vertebral dysplasia in young fish exposed to the herbicide trifluralin. J. Fish Dis. 2: 35–42.

Cranfill, B.E., and A.R. Rhodes. 1987. A comparison of cotton yield response to PPI applications of pendimethalin and trifluralin for weed control. In: Proceedings of the National Cotton Council Beltwide Cotton Production Research Conference, pp. 131–133.

Crosby, D.G., and E. Leitis. 1973. The photodecomposition of trifluralin in water. Bull. Environ. Contam. Toxicol. 10: 237–241.

Dill, T.R., and M.C. Carter. 1973. Preemergence weed control in southeastern forest nurseries. Weed Sci. 21: 363–366.

DiTomaso, J.M. 1988. Effect of herbicides on polyamine formation in roots. Plant Physiol. 86(4, Suppl.): 110 (abstract).

Duseja, D.R. 1982. Soil dissipation of three herbicides. J. Environ. Sci. Health A17: 21–30.

Duseja, D.R., and E.E. Holmes. 1978. Field persistence and movement of trifluralin in two soil types. Soil Sci. 125: 41–48.

Duseja, D.R., H.V. Akunuri, and E.E. Holmes. 1980. Trifluralin soil behaviour: Persistence, movement and weed control. J. Environ. Sci. Health A15: 65–99.

Elmstrom, G.W., and S.J. Locascio. 1974. Evaluation of herbicides for watermelon in Florida. Proc. Fl. State Hortic. Soc. 87: 179–184.

Emmerson, J.L., and R.C. Anderson. 1966. Metabolism of trifluralin in the rat and dog. Toxicol. Appl. Pharmacol. 9: 84–97.

Environment Canada. 1989. Chronic toxicity of lindane, triallate and trifluralin to fathead minnows and *Cenodaphnia dubia*. Report prepared for Environmental Conservation and Protection, Environment Canada, Edmonton, Alberta, by Beak Consultants Ltd.

Eshel, Y., and G.F. Warren. 1966. A simplified method for determining phytotoxicity, leaching, and adsorption of herbicides in soils. Weeds 14: 115–118.

Fabacher, D.L., and H. Chambers. 1974. Resistance to herbicides in insecticide-resistant mosquitofish, *Gambusia affinis*. Environ. Lett. 7: 15–20.

Felix, H.R., R. Chollet, and J. Harr. 1988. Use of the cell wall-less alga *Dunaliella bioculata* in herbicide screening tests. Ann. Appl. Biol. 113: 55–60.

Fenster, C.R., and G.A. Wicks. 1971. Weed control in field beans in Nebraska. Proc. North Cent. Weed Control Conf. 26: 50–52.

Foureman, P.A. 1981a. Identification of aneuploidy inducing chemicals in *Drosophila*. Environ. Mut. 3: 319. (Cited in U.S. EPA 1984.)

Foureman, P.A. 1981b. The TX: A test for the detection of non-disjunction and chromosome breakage in *Drosophila melanogaster*. II. Results of female exposures. Mut. Res. 203: 309–316.

Frank, R., B.D. Ripley, H.E. Braun, B.S. Clegg, R. Johnston, and T.J. O'Neill. 1987. Survey of farm wells for pesticide residues, southern Ontario, Canada, 1981–1982, 1984. Arch. Environ. Contam. Toxicol. 16: 1–8.

Gaynor, J.D. 1985. Dinitroaniline herbicide persistence in soil in southwestern Ontario. Can. J. Soil Sci. 65: 587–592.

Gile, J.D., J.C. Collins, and J.W. Gillet. 1980. Fate of selected herbicides in a terrestrial laboratory microcosm. Environ. Sci. Technol. 14: 1124–1128.

Gjerstad, D.J., and D.B. South. 1981. Preemergence weed control in loblolly, slash, shortleaf, and eastern white pine nursery seedbeds. Can. J. For. Res. 11: 1124–1128.

Glaze, N.C. 1970. Evaluation of herbicides on southern peas. Proc. South. Weed Sci. Soc. 23: 192–196.

Glaze, N.C. 1971. Herbicide evaluation on lima beans. Proc. South. Weed Sci. Soc. 24: 206–211.

Glotfelty, D.W., A.W. Taylor, B.C. Turner, and W.H. Zoller. 1984. Volatilization of surface-applied pesticides from fallow soil. J. Agric. Food Chem. 32: 638–643.

Golab, T., and M.E. Amundson. 1975. Degradation of trifluralin, oryzalin, and isopropalin in soil. In: Environmental quality and safety (Suppl.). Pesticides, ed. F. Coulston and F. Korte. 3: 258–261.

Golab, T., W.A. Althaus, and H.L. Wooten. 1979. Fate of (^{14}C)-trifluralin in soil. J. Agric. Food Chem. 27: 163–179.

Golab, T., R.J. Herber, E.W. Day, A.P. Raun, F.J. Holzer, and G.W. Probst. 1969. Fate of carbon-14 trifluralin in artificial rumen fluid and in ruminant animals. J. Agric. Food Chem. 17: 576–580.

Grande, J.A., and T. Ombrello. 1975. Weed control in transplanted tomatoes with metribuzin in combination with other herbicides. Proc. North. Weed Sci. Soc. 29: 219–224.

Grover, R. 1974. Adsorption and desorption of trifluralin, triallate, and diallate by various adsorbents. Weed Sci. 22(4): 405–408.

Grover, R. 1983. Transport of wild oat herbicides in the environment. In: Wild Oat Symposium Proceedings, Oct. 18–19, 1983, ed. A.E. Smith, pp. 119–132.

Grover, R., and L.A. Kerr. 1981. Evaluation of polyurethane foam as a trapping medium for herbicide vapor in air monitoring and worker inhalation studies. J. Environ. Sci. Health 16: 59–66.

Grover, R., L.A. Kerr, K.E. Bowren and S.U. Khan. 1988. Airborne residues of triallate and trifluralin in Saskatchewan. Bull. Environ. Contam. Toxicol. 40: 683–688.

Grover, R., A.E. Smith, S.R. Shewchuk, A.J. Cessna, and J.H. Hunter. 1988. Fate of trifluralin and triallate applied as a mixture to a wheat field. J. Environ. Qual. 17(4): 543–550.

Grover, R.A.J., J.D. Banting, and P.M. Morse. 1979. Adsorption and bioactivity of diallate, triallate and trifluralin. Weed Res. 19: 363–369.

Hagood, E.S., Jr., J.L. Williams, Jr., and T.T. Bauman. 1980. Influence of herbicide injury on the yield potential of soybeans (*Glycine max*). Weed Sci. 28: 40–45.

Hamilton, K.C., and H.F. Arle. 1972. Persistence of herbicides in fallow desert cropland. Weed Sci. 20: 573–576.

Harper, L.A., A.W. White, Jr., R.R. Bruce, A.W. Thomas, and R.A. Leonard. 1976. Soil and microclimate effects on trifluralin volatilization. J. Environ. Qual. 5(3): 236–242.

Hartnett, J.P. 1975. Weed control in soybeans with RH-2512 and RH-2915. Proc. North. Weed Sci. Soc. 29: 4–8.

Hartzler, R.G., R.S. Fawcett, and M.D.K. Owen. 1989. Effects of tillage on trifluralin residue carryover injury to corn (*Zea mays*). Weed Sci. 37: 609–615.

Harvey, R.G. 1973. Relative phytotoxicities of dinitroaniline herbicides. Weed Sci. 21(6): 517–520.

Harvey, R.G., and G.L. Jacques. 1977. Dinitroaniline herbicides for weed control in peas. Weed Sci. 25: 256–259.

Harvey, R.G., E.T. Gritton, and R.E. Doersch. 1972. Effects of selected herbicides on annual weed control and production of processing peas. Agron. J. 64: 812–815.

Hashimoto, Y., and Y. Nishiuchi. 1982. Effects of herbicides on aquatic animals. In: Pesticide chemistry: Human welfare and the environment, ed. J. Miyamoto and P.C. Kearney. Proceedings of the 5th International Congress of Pesticide Chemistry, Kyoto, Japan, 29 August – 4 September 1982.

Hatfield, H.H., D.T. Warholic, and R.D. Sweet. 1978. Dinitroaniline toxicity to galinsoga, ragweed, and several crops. Proc. North. Weed Sci. Soc. 32: 141–150.

Hayden, B.J., and A.E. Smith. 1980. Comparison of the persistence of ethalfluralin and trifluralin in Saskatchewan field soils. Bull. Environ. Contam. Toxicol. 25: 508–511.

Health and Welfare Canada. 1987. Guidelines for Canadian drinking water quality. 4th ed. Prepared by the Federal-Provincial Subcommittee on Drinking Water of the Federal-Provincial Advisory Committee on Environmental and Occupational Health. Canadian Government Publishing Centre, Ottawa, Ontario. 25 pp.

Helling, C.S. 1971. Pesticide mobility in soils. I. Parameters of thin-layer chromatography. Soil Sci. Soc. Am. Proc. 35: 732–737.

Helling, C.S., and B.C. Turner. 1968. Pesticide mobility: Determination by soil thin-layer chromatography. Science 162: 562.

Henne, R.C. 1977. New compounds with potential for weed control in tomatoes. Proc. North. Weed Sci. Soc. 31: 207–214.

Hertel, C., and D. Marmé. 1983. Herbicides and fungicides inhibit Ca^{2+} uptake by plant mitochondria: A possible mechanism of action. Pestic. Biochem. Physiol. 19: 282–290.

Hess, F.D. 1980. A *Chlamydomonas* algal bioassay for detecting growth inhibitor herbicides. Weed Sci. 28(5): 515–520.

Hoffman, D.J., and P.H. Albers. 1984. Evaluation of potential embryotoxicity and teratogenicity of 42 herbicides, insecticides, and petroleum contaminants to mallard eggs. Arch. Environ. Contam. Toxicol. 13: 15–27.

Hollist, R.L., and C.L. Foy. 1971. Trifluralin interactions with soil constituents. Weed Sci. 19: 11–6.

Horowitz, M., N. Hulin, and T. Blumenfeld. 1974. Behavior and persistence of trifluralin in soil. Weed Res. 14: 213–220.

Huckins, J.N., J.D. Petty, and D.C. England. 1986. Distribution and impact of trifluralin, atrazine, and fonofos residues in microcosms simulating a northern prairie wetland. Chemosphere 15(5): 563-588.

Ivany, J.A., and J.A. Cutcliffe. 1973. Herbicides for cole crops in eastern Canada. Proc. North. Weed Sci. Soc. 27: 194-198.

Jackson, W.T., and D.A. Stetler. 1973. Regulations of mitosis IV. An *in vitro* and ultrastructural study of effects of trifluralin. Can. J. Bot. 51: 1513.

Jensen, K.I.N., and E.R. Kimball. 1980. Persistence of dinitramine and trifluralin in Nova Scotia, Canada. Bull. Environ. Contam. Toxicol. 24: 238-243.

Jensen, K.I.N., J.A. Ivany, and E.R. Kimball. 1983. Efficacy of three dinitroaniline herbicides in processing peas and their residues in soils. Can. J. Plant Sci. 63: 687-693.

Johnson, B.J. 1971. Soybean varieties react similarly to herbicides. G. Agric. Res. 13: 7-8.

Johnson, B.T. 1986. Potential impact of selected agricultural chemical contaminants on a northern prairie wetland: A microcosm evaluation. Environ. Toxicol. Chem. 5: 473-485.

Johnson, W.W., and M.T. Findley. 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. Summaries of toxicity tests conducted at Columbia National Fisheries Research Laboratory, 1965-1978. Resource Publication 137. USDI Fish and Wildlife Service, Washington, D.C.

Jordan, T.N., R.S. Baker, and W.L. Barrentine. 1978. Comparative toxicity of several dinitroaniline herbicides. Weed Sci. 26(1): 72-75.

Junk, G.A., J.J. Richard, and P.A. Dahm. 1984. Degradation of pesticides in controlled water-soil systems. In: Treatment and disposal of pesticide wastes, ed. R.F. Krueger and J.N. Seiber. Am. Chem. Soc. Symp. Ser. 259. American Chemical Society, Washington, D.C.

Karickhoff, S.W., and K.R. Morris. 1985. Impact of tubificid oligochaetes on pollutant transport in bottom sediments. Environ. Sci. Technol. 19: 51-56.

Kearney, P.C., A.R. Isensee, and A. Kontson. 1977. Distribution of dinitroaniline herbicides in an aquatic ecosystem. Pestic. Biochem. Physiol. 7: 242-248.

Ketchersid, M.L., R.W. Bovey, and M.G. Merkle. 1969. The detection of trifluralin vapours in air. Weed Sci. 17: 484-485.

Kosinski, R.J. 1984. The effect of terrestrial herbicides on the community structure of stream periphyton. Environ. Pollut. (Ser. A) 36: 165-189.

Kosinski, R.J., and M.G. Merkle. 1984. The effect of four terrestrial herbicides on the productivity of artificial stream algal communities. J. Environ. Qual. 13(1): 75-82.

Koskinen, W.C., J.E. Oliver, C.G. McWhorter, and P.C. Kearney. 1986. Effect of trifluralin soil metabolities on soybean (*Glycine max*) growth and yield. Weed Sci. 34: 471-473.

Lange, A., D. May, B. Fischer, V. Schweers, and F. Ashton. 1968. Weed control in cucurbits. Calif. Agric. 22: 8-9.

Lange, A.H., B. Fischer, H. Agamalian, and H. Kempen. 1969. Weed control in young grapes. Calif. Agric. 23: 11-12.

LeBaron, H.M., R.R. Wilson, and T.D. Taylor. 1971. Evaluation of CGA-10832 and other herbicides for soybeans in the north central states. Proc. North Cent. Weed Control. Conf. 26: 44-48.

Lee, H.-B., and A.S.Y. Chau. 1983. Determination of trifluralin, diallate, triallate, atrazine, barban diclofop-methyl, and benzoylprop-ethyl in natural waters at parts per trillion levels. J. Assoc. Off. Anal. Chem. 66:651-658.

Leonard, R.A., G.W. Langdale, and W.G. Fleming. 1979. Herbicide runoff from upland piedmont watersheds—Data and implications for modeling pesticide transport. J. Environ. Qual. 8(2): 223-229.

Lignowski, E.M., and E.G. Scott. 1971. Trifluralin and root growth. Plant Cell Physiol. 12: 701-708.

Lignowski, E.M., and E.G. Scott. 1972. Effect of trifluralin on mitosis. Weed Sci. 10: 267.

Liu, D.H.W., and J.M. Lee. 1975. Toxicity of selected pesticides to the bay mussel (*Mytilus edulis*). EPA 660/3-75-016.

Lutman, P.J.W. 1977. Studies on the control of groundkeeper potatoes with the soil-applied herbicides chlorpropham, propyzamide and trifluralin. Pestic. Sci. 8: 637-646.

Macek, K.J., C. Hutchinson, and O.B. Cope. 1969. The effects of temperature on the susceptibility of bluegills and rainbow trout to selected pesticides. Bull. Environ. Contam. Toxicol. 4(3): 174-183.

Macek, K.J., M.A. Lindberg, S. Sauter, K.S. Buxton, and P.A. Costa. 1976. Toxicity of four pesticides to water fleas and fathead minnows. Acute and chronic toxicity of acrolein, heptachlor, endosulfan, and trifluralin to the water flea (*Daphnia magna*) and the fathead minnow (*Pimephales promelas*). EPA/600/3-76-099. 58 pp.

Maguire, R.J., J.H. Carey, M.E. Fox, B.G. Brownlee, and H.-B. Lee. 1988. Assessment of the environmental dynamics of selected pesticides of importance in western Canada. NWRI Contribution #88-20. National Water Research Institute, Canada Centre for Inland Waters, Burlington, Ontario.

Marr, C., and H.D. Swingle. 1970. The effect of several herbicides on weed control and on the element content, yield, and maturity of snap beans (*Phaseolus vulgaris* L.). Proc. South. Weed Sci. Soc. 23: 181-186.

Mayer, F.L., Jr., and M.R. Ellersieck. 1986. Manual of acute toxicity: Interpretation and data base for 410 chemicals and 66 species of freshwater animals. Resource Publication 160. U.S. Department of the Interior, Fish and Wildlife Service.

McBride, R.K., and B.D. Richards. 1975. The effects of some herbicides and pesticides on sodium uptake by isolated perfused gills from the carp (*Cyprinus carpio*). Comp. Biochem. Physiol. 51C: 105-109.

McCorkle, F.M., J.E. Chambers, and J.D. Yarbrough. 1977. Acute toxicities of selected herbicides to fingerling channel catfish, *Ictalurus punctatus*. Bull. Environ. Contam. Toxicol. 18(3): 267-270.

McNevin, G.R., and R.G. Harvey. 1982. Wild proso millet (*Panicum miliaceum*) control in processing peas (*Pisum sativum*) and soybeans (*Glycine max*). Weed Sci. 30: 365-368.

Means, J.C., R.D. Wijayaratne, and W.R. Boynton. 1983. Fate and transport of selected herbicides in an estuarine environment. Can. J. Fish. Aquat. Sci. 40(Suppl 2): 337-345.

Menges, R.M., and T.L. Hubbard. 1970. Selectivity, movement, and persistence of soil-incorporated herbicides in carrot planting. Weed Sci. 18: 247-252.

Merezhinskii, Y.G., and S.V. Sharmankin. 1986. Assembly of microtubules *in vitro* in the presence of trifluralin. Fiziol. Biokhim. Kul't. Rast. 18(3): 299-300 (abstract).

Miller, J.H., L.M. Carter, and C.H. Carter. 1983. An experimental incorporator-planter for cotton (*Gossypium hirsutum*). Weed Sci. 31: 208-214.

Miller, J.H., P.E. Keeley, R.J. Thullen, and C.H. Carter. 1978. Persistence and movement of ten herbicides in soil. Weed Sci. 26: 20-27.

Mitchell, G.A., and F.M. Bourland. 1986. Effects of trifluralin and pendimethalin on cotton emergence and seedling characteristics. In: Proceedings of the National Cotton Council Beltwide Cotton Production Research Conference, pp. 64-66.

Moilanen, K.W., and D.G. Crosby. 1975. Pesticide photo-oxidation in the atmosphere. In: Environmental quality and safety (Suppl.). Pesticides, ed. F. Coulston and F. Korte. 3: 308-312.

Mongar, K., and G.C. Miller. 1988. Vapor phase photolysis of trifluralin in an outdoor chamber. Chemosphere 17(11): 2183-2188.

Moomaw, R.S., and A.R. Martin. 1978. Interaction of metribuzin and trifluralin with soil type on soybean (*Glycine max*) growth. Weed Sci. 26: 327-331.

Moreland, D.E., and S.C. Huber. 1979. Inhibition of photosynthesis and respiration by substituted 2,6-dinitroaniline herbicides. III.

Effects on electron transport and membrane properties of isolated mung bean mitochondria. Pestic. Biochem. Physiol. 11: 247–257.

Moreland, D.E., S.S. Malhotra, R.D. Gruenhagen, and E.H. Shokrah. 1969. Effects of herbicides on RNA and protein synthesis. Weed Sci. 17: 556–563.

Mostafa, I.Y., S.M.A.D. Zayed, Y.M. Adam, and H.S.H. Attaby. 1982. Investigations on trifluralin binding to soil and possible uptake of bound residues by plants. J. Environ. Sci. Health B17: 265–275.

Moyer, J.R. 1979. Soil organic matter, moisture, and temperature effect on wild oats control with trifluralin. Can. J. Plant Sci. 59: 763–768.

Muir, D.C.G., and N.P. Grift. 1987. Herbicide levels in rivers draining two prairie agricultural watersheds (1984). J. Environ. Sci. Health B22: 259–284.

Murnik, M.R. 1978. Mutagenicity of the herbicide trifluralin in *Drosophila melanogaster*. Mut. Res. 53: 235–236 (abstract).

Murphy, H.J., and M.J. Goven. 1976. Weed control in potatoes with several pre-plant incorporated, preemergence, and early post-emergence herbicides. Proc. North. Weed Sci. Soc. 30: 235–238.

Murry, D.S., P.W. Santelmann, and H.A.L. Greer. 1973. Differential phytotoxicity of several dinitroaniline herbicides. Agron. J. 65: 34–36.

Murry, D.S., J.E. Street, J.K. Soteres, and G.A. Buchanan. 1979. Growth inhibition of cotton (*Gossypium hirsutum*) and soybean (*Glycine max*) roots and shoots by three dinitroaniline herbicides. Weed Sci. 27: 336–342.

Naqvi, S.M., and T.S. Leung. 1983. Trifluralin and oryzalin herbicides toxicities to juvenile crawfish (*Procambarus clarkii*) and mosquitofish (*Gambusia affinis*). Bull. Environ. Pollut. 48: 275–283.

Naqvi, S.M., R. Hawkins, and N.H. Naqvi. 1987. Mortality response and LC_{50} values to juvenile and adult crayfish, *Procambarus clarkii*, exposed to Thiodan (insecticide) Treflan, MSMA, Oust (herbicides) and cutrine-plus (algicide). Environ. Pollut. 48: 275–283.

Nash, R.G., and T.J. Gish. 1989. Halogenated pesticide volatilization and dissipation from soil under controlled conditions. Chemosphere 18(11/12): 2353–2362.

Negi, N.S., H.H. Funderburk, Jr., D.P. Schultz, and D.E. Davis. 1968. Effect of trifluralin and nitralin on mitochondrial activities. Weed Sci. 16: 83–85.

Nehez, M., A. Paldy, A. Selypes, and G. Berencsi. 1980. Experiments on the mutagenic effect of two pesticides, DNOC and trifluralin. Mut. Res. 74: 202–203 (abstract).

Nehez, M., A. Selypes, A. Paldy, E. Mazzag, G. Berencsi, and R. Jarmay. 1981. Histopathological and cytogenetical picture of microtoxicological effect of 2 nitrophenol-structured pesticides in mouse experiments on germ cells. Mut. Res. 85(4): 256 (abstract).

Nelson, J.O., P.C. Kearney, J.R. Plimmer, and R.E. Menzer. 1977. Metabolism of trifluralin, profluralin, and fluchloralin by rat liver microsomes. Pestic. Biochem. Physiol. 7: 73–82.

Newsome, L.D., R.L. Lipnick, and D.E. Johnson. 1984. Validation of fish toxicity QSARs for certain non-reactive, non-electrolyte organic compounds. In: QSAR in environmental toxicology, ed. K.L. Kaiser, pp. 279–299. Hingham, Mass.: Kluwer Academic Publication.

Noll, C.J. 1975. Weeding of carrots using preplant incorporation herbicides followed by preemergence and/or postemergent herbicides. Proc. North. Weed Sci. Soc. 29: 244–246.

Nyffeler, A., H.R. Gerber, K. Hurle, W. Pestemer, and R.R. Schmidt. 1982. Collaborative studies of dose-response curves obtained with different bioassay methods for soil-applied herbicides. Weed Res. 22: 213–222.

Oliver, J.E. 1979. Volatilization of some herbicide-related nitrosamines from soils. J. Environ. Qual. 8: 596–601.

Ontario Ministry of Agriculture and Food. 1989. 1990 Guide to weed control. Publication 75. RV-11-89-62M. Queen's Printer for Ontario, Toronto. 208 pp.

O'Sullivan, P.A., and G.N. Prendeville. 1974. Studies on the rate of root growth of intact seedlings in a herbicide medium. Weed Res. 14: 345–348.

Parka, S.J., and J.B. Tepe. 1969. The disappearence of trifluralin from field soils. Weed Sci. 17: 119–122.

Parka, S.J., and H.M. Worth. 1965. The effects of trifluralin to fish. Proc. South. Weed Conf. 18: 469.

Parochetti, J.V. 1975. Weed control in soybeans with metribuzin and combinations with other herbicides. Proc. North. Weed Sci. Soc. 29: 28–35.

Parochetti, J.V., and G.W. Dec. 1978. Photodecomposition of eleven dinitroaniline herbicides. Weed Sci. 26: 153–156.

Parr, J.F., and S. Smith. 1973. Degradation of trifluralin under laboratory conditions of soil anaerobiosis. Soil Sci. 115(1): 55–63.

Pchajek, D.A., I.N. Morrison, and G.R.B. Webster. 1983. Comparison of the efficacy and soil concentrations of fall- and spring-applied trifluralin in flax. Can. J. Plant Sci. 63: 1031–1038.

Plimmer, J.R. 1978. Photolysis of TCDD and trifluralin on silica and soil. Bull. Environ. Contam. Toxicol. 20: 87–92.

Probst, G.W., T. Golab, and W.L. Wright. 1975. Dinitroanilines. In: Herbicides: Chemistry, degradation, and mode of action, 2nd ed. Vol. 1., ed. P.C. Kearney and D.D. Kaufman, pp. 453–500. New York: Marcel Dekker, Inc.

Poe, R.R., D.P. Coyne, B.A. Swisher, and M.D. Clegg. 1988. Differential *Cucurbita* spp. tolerance to the herbicide trifluralin. J. Am. Soc. Hortic. Sci. 113(1): 35–40.

Reeves, S.A., Jr. 1977. Evaluations of selected pre- and post-emergence chemicals on weed control and phytotoxicity to eight sugarcane varieties. Proc. South. Weed Sci. Soc. 30: 130–132.

Reyes, C.C., and R.L. Zimdahl. 1989. Mathematical description of trifluralin persistence in soil. Weed Sci. 37: 604–608.

Rhode, W.A., L.E. Asmussen, E.W. Hauser, R.D. Wauchope, and H.D. Allison. 1980. Trifluralin movement in runoff from a small agricultural watershed. J. Environ. Qual. 9: 37–42.

Rizk, T.Y. 1973. Anatomical and cytological changes in excised cotton roots in response to trifluralin or nitralin treatments. Egypt. J. Bot. 16: 281–289.

Roberts, H.A. 1972. Weed control in mini-cauliflowers. Hortic. Res. 12: 65–71.

Roggenbuck, F.C., and D. Penner. 1987. Factors influencing corn (*Zea mays*) tolerance to trifluralin. Weed Sci. 35: 89–94.

Sanders, H.O. 1970. Toxicities of some herbicides to six species of freshwater crustaceans. J. Water Pollut. Control Fed. 24: 1544–1550.

Sanders, H.O., and O.B. Cope. 1966. Toxicities of several pesticides to two species of cladocerans. Trans. Am. Fish. Soc. 95: 165–169.

Sanders, P.F., and J.N. Seiber. 1983. A chamber for measuring volatilization of pesticides from model soil and water disposal systems. Chemosphere 12(7/8): 999–1093.

Sanok, W.J. 1974. Evaluation of herbicides for control of weeds in Long Island potatoes—1973. Proc. North. Weed Sci. Soc. 28: 282–286.

Savage, K.E. 1978. Persistence of several dinitroaniline herbicides as affected by depth of soil incorporation. Weed Sci. 26: 465–471.

Savage, K.E., and W.L. Barrentine. 1969. Trifluralin persistence as affected by depth of soil incorporation. Weed Sci. 17: 349–352.

Schultz, D.P., H.H. Funderburk, Jr., and N.S. Negi. 1968. Effect of trifluralin on growth, morphology, and nucleic acid synthesis. Plant Physiol. 43: 265–273.

Selleck, G.W., and W.J. Sanok. 1976. Herbicide trial for direct seeded fall cabbage. Proc. North. Weed Sci. Soc. 30: 256–260.

Selleck, G.W., and W.J. Sanok. 1977. Herbicides for weed control in sweet corn and cabbage. Proc. North. Weed Sci. Soc. 31: 256–260.

Sentein, P. 1977. An inhibitor of the achromatic apparatus which alters chromosomes, trifluralin. Arch. Anat. Microsc. 66(4): 263–277.

Sheehan, P.J., R.P. Axler, and R.C. Newhook. 1986. Evaluation of simple generic aquatic ecosystem tests to screen the ecological impacts of pesticides. In: Community toxicity testing, ed. J. Cairns, Jr., pp. 158–179. American Society for Testing and Materials, ASTM STP 920.

Sheets, T.J., J.R. Bradley, Jr., and M.D. Jackson. 1973. Movement of trifluralin in surface water. Proc. South. Weed Sci. Soc. 26: 376 (abstract).

Smith, A.E. 1975. Field persistence studies with herbicides in prairie soils. In: Environmental quality and safety (Suppl.). Pesticides, ed. F. Coulston and F. Korte, 3: 266–270.

Smith, A.E. 1979. Soil persistence experiments with (^{14}C)-2, 4-D in herbicidal mixtures, and field persistence studies with triallate and trifluralin both singly and combined. Weed Res. 19: 165–170.

Smith, A.E. 1983. Persistence of wild oat herbicides. In: Wild Oat Symposium Proceedings, Oct. 18–19, 1983, ed. A.E. Smith, Agriculture Canada, pp. 111–116.

Smith, A.E., and K.P. Bucholtz. 1964. Modification of plant transpiration rate with chemicals. Plant Physiol. 39: 572–578.

Smith, A.E., and B.J. Hayden. 1976. Field persistence studies with eight herbicides commonly used in Saskatchewan. Can. J. Plant Sci. 56: 769–771.

Smith, A.E., and B.J. Hayden. 1982a. Field persistence studies with triallate and trifluralin both singly and in combination with chloramben. Bull. Environ. Contam. Toxicol. 29: 240–242.

Smith, A.E., and B.J. Hayden. 1982b. Carry-over of dinitramine, triallate, and trifluralin to the following spring in soils treated at different times during the fall. Bull. Environ. Contam. Toxicol. 29: 483–486.

Smith, A.E., A.J. Aubin, and D.A. Derksen. 1988. Loss of trifluralin from clay and loam soils containing aged and freshly applied residues. Bull. Environ. Contam. Toxicol. 41: 569–573.

Solbakken, E., H. Hole, O. Lode, and T.A. Pedersen. 1982. Trifluralin persistence under two different soil and climatic conditions. Weed Res. 22: 319–328.

South, D.B. 1977. Pre- and postemergent weed control in forest nurseries. Proc. South. Weed Sci. Soc. 30: 269–278.

Spacie, A., and J.L. Hamelink. 1979. Dynamics of trifluralin accumulation in river fishes. Environ. Sci. Technol. 13: 817–822.

Spain, J.C., and P.A. van Veld. 1983. Adaptation of natural microbial communities to degradation of xenobiotic compounds: Effects of concentration, exposure time, inoculum, and chemical structure. Appl. Environ. Microbiol. 45(2): 428–435.

Sparchez, C., V. Soran, C. Craciun, A. Polizu, and V. Craciun. 1987. Trifluralin influence on the ultrastructure of root meristem cells in wheat (*Triticum aestivum* L.) and sunflower (*Hellianthus annuus* L.). Rev. Roum. Biol. Ser. Biol. Veg. 32(1): 39–41.

Spencer, W.F., and M.M. Cliath. 1974. Factors affecting vapour loss of trifluralin from soil. J. Agric. Food Chem. 22(6): 987–991.

Statistics Canada. 1988. Imports. Commodity by country: C.I.T.C. detail (1984–1985 and 1986–1987). International Trade Division.

Struckmeyer, B.E., L.K. Binning, and R.G. Harvey. 1976. Effect of dinitroaniline herbicides in a soil medium on snap bean and soybean. Weed Sci. 24: 366–369.

Sullivan, R.G., H.W. Knoche, and J.C. Markle. 1980. Photolysis of trifluralin: Characterization of azobenzene and azooxybenzene photodegradation products. J. Agric. Food Chem. 28: 746–755.

Suntio, L.R., W.Y. Shiu, D. Mackay, J.N. Seiber, and D. Glotfelty. 1988. Critical review of Henry's law constants for pesticides. Rev. Environ. Contam. Toxicol. 103: 1–56.

Swann, C.W., and R. Behrens. 1972a. Phytotoxicity of trifluralin vapors from soil. Weed Sci. 20: 143–146.

Swann, C.W., and R. Behrens. 1972b. Trifluralin vapor emission from soil. Weed Sci. 20: 147–149.

Taylor, A.W. 1978. Post-application volatilization of pesticides under field conditions. J. Air Pollut. Control Assoc. 28(9): 922–927.

Teasdale, J.R., R.G. Harvey, and D.J Hagedorn. 1978. Suppression of pea (*Pisum sativum*) root rot by dinitroaniline herbicides. Weed Sci. 26: 609–613.

Therrien-Richards, S., and D.A. Williamson. 1987. Contamination by pesticides of the La Salle and Assiniboine rivers, Manitoba, Canada. W&NR 87/88-CP(EP)-1. Environmental Protection Service, Western and Northern Region.

Udoh, D.J., and L.E. Nelson. 1986. Trifluralin-induced chlorosis in soybean (*Glycine max*)(L.) Merr. grown on clayey, high pH soils. Plant Soil 96: 175–184.

U.S. EPA (U.S. Environmental Protection Agency). 1979. Trifluralin (Treflan). Position document 1/2/3. Special Pesticide Review Division, Washington, D.C.

U.S. EPA (U.S. Environmental Protection Agency). 1982. Pesticide rebuttal presumption against registration, Treflan (trade name). Position document 4. Special Pesticide Review Division, Washington, D.C.

U.S. EPA (U.S. Environmental Protection Agency). 1984. Health and environmental effects profile for trifluralin. EPA/600/X-84/234. Environmental Criteria and Assessment Office, Office of Health and Environmental Assessment, Office of Research and Development, Cincinnati, Ohio.

U.S. EPA (U.S. Environmental Protection Agency). 1987a. Guidance for the registration of pesticide products containing trifluralin as the active ingredient. Office of Pesticide Programs, Registration Division, Washington, D.C.

U.S. EPA (U.S. Environmental Protection Agency). 1987b. Trifluralin health advisory. Office of Drinking Water, Washington D.C. Draft report.

Vaughn, M.A., and K.C. Vaughn. 1986. Carrot microtubules are resistant to some dinitroaniline herbicides. J. Cell Biol. 103(5 part 2): 272a (abstract).

Verschueren, K. 1983. Handbook of environmental data on organic chemicals. 2nd ed. New York: Van Nostrand Reinhold Co.

Waite, D., H. Sommerstad, R. Grover, and L.A. Kerr. 1986. South Saskatchewan watershed study (1985 interim report). W&NR 86/87-CP (EPS)-3. Environmental Protection Service, Western and Northern Region, Regina, Sask.

Walker, W.W., C.R. Cripe, P.H. Pritchard, and A.W. Bourquin. 1988. Biological and abiotic degradation of xenobiotic compounds in *in vitro* estuarine water and sediment/water systems. Chemosphere 17(12): 2255–2270.

Warner, J.E., S.R. Winter, and A.F. Wiese. 1987. Persistence of dinitroaniline herbicides and potential for injury to sugar beets. J. Am. Soc. Sugar Beet Technol. 24(1): 57–66.

Wauchope, R.D. 1978. The pesticide content of surface water draining from agricultural fields—A review. J. Environ. Qual. 7: 459–472.

Wauchope, R.D., K.E. Savage, and J.M. Chandler. 1977. Soil sample variation and herbicide incorporation uniformity. Weed Sci. Soc. Am. 25: 193–196.

Weber, J.B. 1987. Physical/chemical interactions of herbicides with soil. Proc. Calif. Weed Conf. 39: 96–109.

Weber, J.B. 1990. Behavior of dinitroaniline herbicides in soils. Weed Technol. 4: 394-406.

Webster, G.R.B., C.F. Shaykewick, S. Kanhai, and G.J. Reimer. 1978. Availability of the herbicide trifluralin for control of wild oats as influenced by soil characteristics in four Manitoba soils. Can. J. Soil Sci. 58: 397–404.

Wells, D.E., and A.A. Cowan. 1982. Vertebral dysplasia in salmonids caused by the herbicide trifluralin. Environ. Pollut. (Ser. A) 29: 249–260.

Wheeler, W.B., G.D. Stratton, R.R. Twilley, L.T. Ou, D.A. Carlson, and J.M. Davidson. 1979. Trifluralin degradation and binding in soil. J. Agric. Food Chem. 27: 702–706.

White, A.W., Jr., L.A. Harper, R.A. Leonard, and J.W. Turnbull. 1977. Trifluralin volatilization losses from a soybean field. J. Environ. Qual. 6(1): 105–110.

WHO (World Health Organization). 1987. Drinking-water quality guidelines for selected herbicides. Copenhagen, Denmark.

Williamson, D.A. 1984. A preliminary investigation into the presence of agricultural pesticides in the La Salle and Assiniboine rivers, Manitoba, Canada. Water Standards and Studies Report No. 84-5. Environmental Management Services Branch, Manitoba Department of Environment and Workplace Safety and Health.

Willis, G.H., R.L. Rogers, and L.M. Southwick. 1975. Losses of diuron, linuron, fenac, and trifluralin in surface drainage water. J. Environ. Qual. 4: 399–402.

Willis, G.H., R.C. Wander, and L.M. Southwick. 1974. Degradation of trifluralin in soil suspensions as related to redox potential. J. Environ. Qual. 3(3): 262–265.

Willis, G.H., L.L. McDowell, C.E. Murphree, L.M. Southwick, and S. Smith, Jr. 1983. Pesticide concentrations and yields in runoff from silty soils in the lower Mississippi valley. J. Agric. Food Chem. 31: 1171–1177.

Windholz, M., S. Budavari, R.F. Blumetti, and E.S. Otterbein. 1983. The Merck index: An encyclopedia of chemicals, drugs, and biologicals. Rahway, N.J.: Merck and Co., Inc.

Wnuk, M., R. Kelley, G. Breuer, and L. Johnson. 1987. Pesticides in water supplies using surface water sources PB 88-136916/XAD. U.S. Environmental Protection Agency. Washington, D.C. 43 pp.

Woodrow, J.E., D.G. Crosby, T. Mast, K.W. Moilanen, and J.N. Seiber. 1978. Rates of transformation of trifluralin and parathion vapors in air. J. Agric. Food Chem. 26: 1312–1316.

Worth, H.J., and R.C. Anderson. 1965. The toxicity of trifluralin, treflan, a herbicide to mammals and chickens. Proc. South. Weed Sci. Soc. 18: 711–712.

Worthing, C.R., and S.B. Walker (eds). 1987. The pesticide manual: A world compendium. 8th ed. British Crop Protection Council, Thornton Heath, U.K.

Wright, W.L., and G.F. Warren. 1965. Photochemical decomposition of trifluralin. Weeds 13: 329–331.

Yockim, R.S., A.R. Isensee, and E.A. Walker. 1980. Behavior of trifluralin in aquatic model ecosystems. Bull. Environ. Contam. Toxicol. 24: 134–141.

Yoon, J.S., J.M. Mason, R. Valencia, R.C. Woodruff, and S. Zimmering. 1985. Chemical mutagenesis testing in *Drosophila*. IV. Results of 45 coded compounds tested for the national toxicology program. Environ. Mutagen. 7: 349–367.

Zayed, S.M.A.D., I.Y. Mostafa, M.M. Farghaly, H.S.H. Ataby, Y.M. Adam, and F.M. Mahdy. 1983. Microbial degradation of trifluralin by *Aspergillus carneus*, *Fusarium oxysporum* and *Trichoderma viride*. J. Environ. Sci. Health B18: 253–267.

Zepp, R.G., and D.M. Cline. 1977. Rates of direct photolysis in aquatic environment. Environ. Sci. Technol. 11(4): 359–366.

Zeyer, J., and P.C. Kearney. 1983. Microbial dealkylation of trifluralin in pure culture. Pestic. Biochem. Physiol. 20: 10–18.

Zilkah, S., P.F. Bocion, and J. Gressel. 1977. Cell cultures vs. whole plants for measuring phytotoxicity. II. Correlations between phytotoxicity in seedlings and calli. Plant Cell Physiol. 18: 657–670.

Zimdahl, R.L., and S.M. Gwynn. 1977. Soil degradation of three dinitroanilines. Weed Sci. 25: 247–251.

Appendix A

Trifluralin Residues in Runoff from Agricultural Land

Table A-1. Trifluralin Residues in Runoff from Agricultural Land

Plot description (soil type, crop)	Formulation (% ai)	Application rate ($kg \cdot ha^{-1}$)	Method of application	Residues in runoff ($mg \cdot L^{-1}$) and days posttreatment	Reference
Bushland, Texas: Pullman silty clay loam (1.6% O.M.); water samples collected in plastic cups (buried in treated farmers' fields) each time it rained	NR	NR	NR	0.04, the highest concentration found in any of the samples; no time frame report, but mentioned analysis of soil samples 3 mo. posttreatment	Axe, Mathers, and Weise 1969
Walkinsville, Georgia: Gravelly sandy loam; 2%–10% slope; 2.71 ha; soybean crop	NR	1.12	NR (sprayed as aqueous emulsion; incorporated into soil)	0.007–0.015 (runoff water plus suspended sediment); in 8–9 runoff events over a 3- to 4-mo. period	Wauchope 1978
Sandy loam/sandy clay; 3% slope; 1.26 ha; soybean crop	NR	1.12	NR (applied as above)	0.009–0.01 (runoff water plus suspended sediment); measured below grass in 4–10 runoff events over a 2-mo. period	
Lewiston and Rocky Mount, North Carolina: Sandy loam and loamy sand; 2% and 4% slope respectively; 0.0017 ha (17-m^2 plots); cotton crop	NR	1.12	NR (applied as above)	0.008–0.024 (runoff water plus suspended sediment); 10–21 runoff events over a 5- to 8-mo. period	
Stoneville, Mississippi: Sandy loam; 0.5% slope; 0.20 ha; soybean crop	NR	0.84	NR (applied as above)	0.0005–0.00027 (runoff water plus suspended sediment); in 2–7 runoff events over a 1- to 5-mo. period	
Clarksdale, Mississippi: Silty loam; 0.2% slope; 15.6 ha					
Soybean crop	NR	1.12	NR (applied as above)	0.18% (2.0 $g \cdot ha^{-1}$) of that applied lost in 21 runoff events over a 12-mo. period	
Cotton crop	NR	1.12	NR (applied as above)	0.18% (2.0 $g \cdot ha^{-1}$) of that applied lost in 39 runoff events over a 12-mo. period	
Georgia: Cecil sandy loam, Pacolet gravelly sandy loam; no soil and water conservation structures present; 2.7 ha; 3% average slope; crop of soybeans; conventionally tilled before planting/no cultivating after plant	NR	1.12 (July 1, 1972)	NR (applied as recommended for crop production; applied with diphenamid at 3.36 $kg \cdot ha^{-1}$)	0–0.006 in solution (dissolved) and 0–0.1 ppm in sediment; in 6 runoff events over one 90-d period (27 d from application to first runoff event)	Leonard, Landgdale, and Fleming 1979

NR = not reported
ND = not detected
O.M. = organic matter

Table A-1. Continued

Plot description (soil type, crop)	Formulation (% ai)	Application rate ($kg \cdot ha^{-1}$)	Method of application	Residues in runoff ($mg \cdot L^{-1}$) and days posttreatment	Reference
As above	NR	1.12 (June 13, 1973)	NR (applied as above)	0–0.013 in solution (dissolved) and 0.01–0.06 ppm in sediment; in 9 runoff events over one 90-d period (<1 d from application to first runoff event)	
Cecil sandy loam; portion of a parallel-terraced area with grassed outlet channels serving to collect runoff; 1.3 ha; 3% average slope; crop of soybeans; conventionally tilled before planting and no cultivating after planting	NR	1.12 (June 30, 1972)	NR (applied as above)	0–0.021 in solution (dissolved) and 0–0.28 ppm in sediment; in 10 runoff events over one 90-d period (2 d from application to first runoff event)	
As above	NR	1.12 (June 15, 1973)	NR (applied as above)	0–0.008 in solution (dissolved) and 0–0.09 ppm in sediment; in 5 runoff events over one 90-d period (23 d from application to first runoff event)	
North Carolina: 8 surface runoff plots (4 treated and 4 controls) each surrounded by sheet metal with a catchment device at plot lower end to collect runoff; cotton planted	NR	1.12	NR (incorporation to 10 cm before planting)	Runoff samples collected after each rain producing considerable runoff. Suction filtered sediments contained an average of 84% of trifluralin detected in the runoff. Highest concentration in surface runoff was 0.024 at one location 6 wk after application in 1970. Concentrations generally higher for first few rains after application and gradually decreased.	Sheets, Bradley, and Jackson 1973
Small pond in a watershed	NR	0.84 to 50%–60% of watershed	NR	Highest concentration was 0.002 $mg \cdot L^{-1}$ 5 d after application.	
Baton Rouge, Louisiana: Mhoon silty clay, 0.93%–1.42% O.M.; pH 5.5–6.0; 0.2% slope; plot rows and small berms used to channel runoff through Parshall flume (measured volume) and Geibmultislot divisor (diverted 1/9 aliquot to storage tank). Planted with cotton and soybeans, but so similar in terms of runoff concentrations that treated as duplicates. Plots 0.045 ha each.	NR	1.4 April 30, 1971, March 29, 1972, April 30, 1973	Aqueous emulsion broadcast with small tractor-mounted sprayer, preplant, and soil incorporated (double disked to 7.5 cm)	Lower limit of accurate quantification 0.01. Ranges provided as a result of data typically as fractions of $\mu g \cdot L^{-1}$. 0%–0.04% of that applied lost in 5–6 runoff events over a 3- to 4-mo. period.	Willis, Rogers, and Southwick 1975

Table A-1. Continued

Plot description (soil type, crop)	Formulation (% ai)	Application rate ($kg \cdot ha^{-1}$)	Method of application	Residues in runoff ($mg \cdot L^{-1}$) and days posttreatment	Reference
Tifton, Georgia: Cowarts loamy sand, 0.5% O.M.; <3% slope; 0.34-ha watershed planted with soybeans (bedded) each year (July 12, 1974, and May 14, 1975) after fertilization with 0-10-20 at 560 $kg \cdot ha^{-1}$	NR	1.12 (ai)	Surface applied 1 d before planting and incorporated to 10-cm depth with a rototiller		Rhode *et al.* 1980
Shallow phreatic flow, above semi-permeable formation (92–214 cm depth), collected by a tile drain at low side of watershed and directed through a V-notch weir and collected by hand (450 mL). The subsurface watershed is 0.36 ha.	NR	July 11, 1974	Applied as above	ND–0.0003 (runoff water plus suspended sediments) in 0–2 runoff events per mo. (8 total) over a 1-yr. period (no runoff after 9th mo.)	
		May 13, 1975	Applied as above	ND (runoff water plus suspended sediments), 0–3 runoff events per mo. (11 total) over a 1-yr. period (no runoff after 7th mo.)	
Surface runoff directed through a grassed waterway adjoining the watershed and through an H-flume at bottom. Samples collected by hand.	NR	July 11, 1974	Applied as above	ND–0.038 (runoff water plus suspended sediments) in 0–4 runoff events per mo. (21 total) over a 1-yr. period (no runoff after 9th mo.)	
		May 13, 1975	Applied as above	ND–0.023 (runoff water plus suspended sediments) in 0–6 runoff events per mo. (30 total) over a 1-yr. period (no runoff after 8th mo.)	
				Runoff was normal or above normal for both years.	
Three separate subplots (28 m^2) within above watershed with simulated rainfall at various days after application (pipes 1.51 m above group applying water at 19.1 $cm \cdot hr^{-1}$ for 30 min). Runoff collected by a gutter at plot edge and directed through an HS-flume for measurement and sampling. Simulated rainfall greater than expected under natural conditions and therefore greater runoff losses of trifluralin expected.		1.12 1974	Applied as above	0.025 + 1 d	
				0.004 + 29 d	
				0.0001 + 71 d	
		1975	Applied as above	0.012 + 10 d	
				0.009 + 21 d	
				0.004 + 38 d	
				Concentration curves for sampling period after each artificial watering integrated with the discharge hydrograph gives total load loss and maximum time-weighted average concentration for each event.	

Table A-1. Continued

Plot description (soil type, crop)	Formulation (% ai)	Application rate (kg ha^{-1})	Method of application	Residues in runoff (mg L^{-1}) and days posttreatment	Reference
Same watershed as above. 2-bed, 4-row subplot with runoff directed onto the grassed waterway (above). Discharge measured and samples collected at two HS-flumes, one at the waterway entrance and one at the outlet. Samples also taken at 3 intermediate sites. Wet waterway (9.55 cm rainfall in 2 wk prior) with dry waterway (no rainfall 2 wk prior); both sprinkled by 1.3 cm water 1 d before test. Subplot artificially watered at same rate and amount of time as above during test on June 11, 1976 (wet) and Oct. 19, 1977 (dry).		1.12 June 10, 1976 (wet)	Applied as above	Upper flume 0.005–0.012 Intermediate max. 0.008 Lower flume 0.001–0.002	
		Oct. 18, 1977	Applied as above	Upper flume 0.006–0.013 Intermediate max. 0.001–0.002 Lower flume <0.004	
Clarksdale, Mississippi: 18.7-ha watershed: mean slope of 0.2% with several soil types: Bruin silt loam (57%), Commerce silt loam (15%), Tunica silty clay (11%), and Sharkey silty clay (17%). Runoff directed by shallow V-ditches into a 1.6-ha pond on the watershed. Before entering the pond 64% of the runoff was measured and sampled (i.e., only one drainage ditch carried sampling instruments). Samples collected with an automatic pump at 10-min intervals throughout each runoff event (storm). Planted to cotton each year and stalks shredded after harvest (fertilized in spring).	NR	1.12 March 1972	NR preplant application		Willis *et al.* 1983
		May 1973		0.0002	
		April 1974		0.0001	
		April 1975		0.0004	
		April 1976		0.0008	
		April 1977		0.0005	
		April 1978		0.0004 The above are values for the year. The concentrations from each storm event are discharge weighted. A range of r^2 values from 0.55 to 0.98 for years 1973–1978 relate storm pesticide yield (g ha^{-1}) to storm sediment yield (kg ha^{-1})	
Outlook, Saskatchewan: Irrigation basin	NR	NR	NR	0.0007 in tailwaters following first irrigation <0.0001 in drain canal (carried all waste water from the irrigation basin)	Grover 1983

Appendix B

Environmental Concentration Ranges of Trifluralin Residues in Surface Water, Groundwater, Sediment, and Biota

Table B-1. Environmental Concentration Ranges of Trifluralin Residues in Surface Water, Groundwater, Sediment, and Biota

Sample	Location, years, and conditions	Matrix	Concentration range (& mean)	Samples with pesticide	Reference
27 samples from 2 locations on 16 occasions	La Salle River, Man. Sampling interval clustered during April 1983 to coincide with snowmelt water run-off and at monthly intervals from May 1983 to March 1984 (excluding August 1983). Sample filtration prior to extraction removed 90% of the 20- to 25-mm size particles. Drains agricultural land.	Water	ND–0.24 $\mu g \cdot L^{-1}$. Highest values from one location on April 18, 1983, and Nov. 8, 1983, and other values typically ND or trace amounts. Detection limit: 0.05 $\mu g \cdot L^{-1}$	NR	Williamson 1984
15 samples from 2 sites on 10 occasions	Assiniboine River, Man. Sampling at monthly intervals from May 1983 to March 1984 (excluding August 1983). Drains agricultural land.		ND–0.1 $\mu g \cdot L^{-1}$. Highest value from one location on Nov. 8, 1983, and other values typically ND or trace amounts. Detection limit: 0.05 $\mu g \cdot L^{-1}$	NR	
Single samples collected on 1 occasion from 3 small water pools within study area	April 11, 1983		Trace	NR	
	June 1, 1983		Trace	NR	
	June 1, 1983		ND (detection limit: NR)	NR	
NR	NR	Fish tissue	0.775 $mg \cdot kg^{-1}$ (trifluralin)	NR	
			1.323 $mg \cdot kg^{-1}$ (trifluralin & metabolites)	NR	
NR	NR	Snail tissue	27.085 $mg \cdot kg^{-1}$ (trifluralin)	NR	
			28.870 $mg \cdot kg^{-1}$ (trifluralin & metabolites)	NR	
1 location	Ochre River, West. Man. 3.5-L grab sampling in duplicate using 4-L amber glass bottles on March 14, April 13, April 27, and at weekly intervals afterward until Sept. 5, 1984. Final collection on Oct. 10, 1984. Drains mainly non-cropped land and forest.	Water	Maximum levels did not exceed 25×10^{-3} $\mu g \cdot L^{-1}$, and found at detectable levels ($<3 \times 10^{-3}$ $\mu g \cdot L^{-1}$) on only three occasions (May, June, July) or 10% of the samples.	NR	Muir and Grift 1987
	Turtle River. As above. Drains mainly agricultural land.	Water	Maximum levels did not exceed 25×10^{-3} $\mu g \cdot L^{-1}$; below detection limits (3.5×10^{-3} $\mu g \cdot L^{-1}$) in almost all samples (exceeded 3×10^{-3} $\mu g \cdot L^{-1}$ in 14% of the samples).	NR	

NR = not reported
ND = not detected

Table B-1. Continued

Sample	Location, years, and conditions	Matrix	Concentration range (& mean)	Samples with pesticide	Reference
7 sampling locations along length of river	LaSalle River, Man. One grab sample per site at 30-day intervals from Aug. to Dec. 1984 with a 1-L Boston round bottle at midstream. Drains agricultural land.	Water	ND (detection limit: 0.1 $\mu g \cdot L^{-1}$). Possibly not detected because during May (usual month of trifluralin application) rainfall below normal.	NR	Therrien-Richards and Williamson 1987
As above	Sampling with Ekman dredge at 3 equidistant points across stream width at each sampling location on 1 occasion in Aug. 1984 (1 sample per sampling site).	Sediment	0.004 $mg \cdot kg^{-1}$	1/21	
4 of the above 7 sampling sites and 3 subsamples at one site for a total of 6 samples	Samples of small forage fish collected by seine, nets, and basket-type minnow traps. Sampling data is variable (some given, some not). Samples equal 100 g of each fish species.	Fish tissue: brown bullhead (*Ictalurus nebulosus*)	0.0045–0.0057 $mg \cdot kg^{-1}$ (0.0049 $mg \cdot kg^{-1}$)	NR	
		brook stickleback (*Culaea inconstans*)	0.0047 $mg \cdot kg^{-1}$	NR	
		central mudminnow (*Umbra limi*)	ND–0.0075 $mg \cdot kg^{-1}$ (detection limit: NR)	NR	
Sample 4 of above 7 sampling sites	100 g sampled from each site	Aquatic macrophytes	NR	NR	
2 sampling sites along river	Assiniboine River, Man. (downstream Trans-Canada Highway). One midstream grab sample per site at 30-d intervals from Aug. to Dec. 1984 with a 1-L bottle. Drains agricultural land.	Water	ND (detection limit: 0.1 $\mu g \cdot L^{-1}$). Possibly not detected because during May (usual month of trifluralin application) rainfall below normal.	NR	
As above	Sampling by hand of fine-grained deposits on lee side of midstream obstructions (sand bars and rocks) on 1 occasion in Aug. 1984. Number of samples NR.	Sediment	0.006 $mg \cdot kg^{-1}$ (1 sample)	NR	
NR	NR	River water	1.8 $\mu g \cdot L^{-1}$	NR	
		Whole fish tissue	3.24–10.78 $mg \cdot kg^{-1}$	NR	
7 sampling sites	Study area 2800 ha operated by 17 farmers and the City of Regina. Sampling on a daily basis for duration of runoff event at 4 culverts crossing into study area at a stream connecting 2 permanent sloughs and	Spring runoff	ND (detection limit: 0.1 $\mu g \cdot L^{-1}$)	NR	Waite *et al.* 1986

Table B-1. Continued

Sample	Location, years, and conditions	Matrix	Concentration range (& mean)	Samples with pesticide	Reference
	at a culvert siting the lower slough. Only 2 runoff events reported (March 27 and 28, 1984) apparently because of small snowfall and cold spring. Grab samples collected in 4-L glass bottles.				
35 sampling sites, one sample from each site.	Iowa. Samples collected after rainfall from 35 treated public water supplies between May 1 and July 1, 1986, using 2 one-quart glass jars. 33 water supply samples analyzed.	Surface water	0.13 $\mu g\ L^{-1}$	1/33	Wnuk *et al.* 1987
14 of the above 33 sampling sites, plus 1 other sampling site; 1 sample at each site	Samples collected after rainfall from 15 untreated public water supplies (not including the one site where trifluralin was detected in treated water) between May 1 and July 1, 1986, using same sample size as above.	Surface water	ND (detection limit: 0.1 $\mu g\ L^{-1}$)	0/15	
45 samples (number of sampling sites NR)	University of Iowa. Samples collected approx. weekly, May 1985 to March 1986 from untreated community public water supply systems.	Surface water	ND (detection limit: 0.2 $\mu g\ L^{-1}$)	0/45	
48 samples (number of sampling sites NR)	As above - treated water	Surface water	ND (detection limit: 0.2 $\mu g\ L^{-1}$)	0/48	
44 samples (number of sampling sites NR)	Davenport, Iowa. As above - untreated water	Surface water	ND (detection limit: 0.2 $\mu g\ L^{-1}$)	0/44	
46 samples (number of sampling sites NR)	As above - treated water	Surface water	ND (detection limit: 0.2 $\mu g\ L^{-1}$)	0/46	
41 samples (number of sampling sites NR)	Clarinda, Iowa. As above - untreated water	Surface water	ND (detection limit: 0.2 $\mu g\ L^{-1}$)	0/41	
46 samples (number of sampling sites NR)	As above - treated water	Surface water	ND (detection limit: 0.2 $\mu g\ L^{-1}$)	0/46	

Table B-1. Continued

Sample	Location, years, and conditions	Matrix	Concentration range (& mean)	Samples with pesticide	Reference
15 samples of 2 fish species from each of 14 sampling sites	Lake Michigan, Michigan. Sample sites included 13 tributaries and 1 bay of Lake Michigan. Samples collected in fall 1983. Each species separated according to size (assumed to positively reflect age and therefore bioconcentration) and group with largest fish analyzed.	Whole fish homogenate (20-g samples)			Camanzo *et al.* 1987
		common carp	0.003–0.126 $mg \cdot kg^{-1}$ (0.028 $mg \cdot kg^{-1}$)	NR	
		smallmouth bass (*Micropterus dolomieui*)	0.005–0.011 $mg \cdot kg^{-1}$ (0.008 $mg \cdot kg^{-1}$)	NR	
		channel catfish (*Ictalurus punctatus*)	0.050 $mg \cdot kg^{-1}$	NR	
		pumpkinseed (*Lepomis gibbosus*)	0.004 $mg \cdot kg^{-1}$	NR	
		bowfin (*Amia calva*)	0.018–0.034 $mg \cdot kg^{-1}$ (0.024 $mg \cdot kg^{-1}$)	NR	
		northern pike (*Esox lucius*)	0.004–0.100 $mg \cdot kg^{-1}$ (0.029 $mg \cdot kg^{-1}$)	NR	
		rock bass (*Ambloplites rupestris*)	ND–0.008 $mg \cdot kg^{-1}$ (0.021 $mg \cdot kg^{-1}$) Detection limit: 0.003 $mg \cdot kg^{-1}$	NR	
		lake trout (*Salvelinus namaycush*)	0.011 $mg \cdot kg^{-1}$	NR	
		largemouth bass (*Micropterus salmoides*)	0.011 $mg \cdot kg^{-1}$	NR	
15 fish from each of 12 sampling sites. 3 samples per site - each comprising 5 single fillets (skin on) from each fish	Fall 1980. 12 sites throughout all the Great Lakes. Adult fish collected on tributaries as they began their fall upstream migration. 36 samples total.	Fillet homogenate coho salmon (*Oncorhynchus kisutch*)	ND in any samples (detection limit: 0.005 $mg \cdot kg^{-1}$) except 1, which produced a trace of trifluralin (present but below 0.01 $mg \cdot kg^{-1}$)	1/36	Clark, DeVault, and Bowden 1984
2 samples per 91 sample sites	Nov. 25 and Dec. 16, 1984; 91 farm wells across southern Ontario sampled on these two dates; 1.5-L samples. Trifluralin detected in only 1 well (13 m deep, sandy soil, used for mixing and filling tanks).	Groundwater	0.041 $mg \cdot L^{-1}$	NR	Frank *et al.* 1987

Appendix C

Summary of Studies of Trifluralin Persistence in Soil

Table C-1. Summary of Studies of Trifluralin Persistence in Soil

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues ($mg \cdot kg^{-1}$ except when noted)	Results and comments	Reference
Melfort, Sask.: Melfort silty clay (11.7% O.M., pH 5.2; 36% field capacity moisture)	1.1 $kg \cdot ha^{-1}$	0–5 cm	% of applied chemical remaining (mean ± SD) 71 ± 4 (Oct. 71–May 72) 55 ± 1 (Oct. 72–May 73) 31 ± 7 (May 72–Oct. 72) 24 ± 6 (May 73–Oct. 73) 35 ± 3 (May 72–May 73) 16 ± 3 (May 72–Oct. 73)	Incorporated into the top 5 cm of soil for 2 min with a fork to reduce volatility. Plots tamped down to prevent wind erosion and weeded regularly with minimal disturbance. Considerably milder winter of 1972–73 may have contributed to decreased carry-over (soil temperature and moisture content prior to freeze-up and after spring thaw may have contributed to losses by biological degradation and volatilization). Increased rainfall was recorded during the 1973 studies compared to the 1972 studies (23.9 and 47.2 cm as compared to 19.0 and 18.5 cm for Regina heavy clay and Melfort silty clay respectively) and was considered a possible contributing factor to lower residue levels in Oct. 1973 than in Oct. 1972.	Smith 1975
		5–10 cm	Negligible residues (<2%)		
Regina, Sask.: Regina heavy clay (4.2% O.M.; pH 7.7; 40% field capacity moisture)	1.1 $kg \cdot ha^{-1}$	0–5 cm	71 ± 11 (Oct. 71–May 72) 32 ± 8 (Oct. 72–May 73) 12 ± 4 (May 72–Oct. 72) 8 ± 2 (May 73–Oct. 73) 16 ± 5 (May 72–May 73) 3 ± 0 (May 72–Oct. 73)		
		5–10 cm	Negligible residues (<2%)		

NR = not reported
O.M. = organic matter
RMS = regression mean square

Note: Field capacity is interpreted as the % soil moisture (by weight) retained by a saturated soil after it has been allowed to drain by gravity for three days (x% field capacity or field capacity x% indicates a soil's field capacity; x% of field capacity indicates the existing soil moisture as a fraction of field capacity).

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues ($mg \cdot kg^{-1}$ except when noted)	Results and comment	Reference
Jameson, Sask.: Jameson sandy loam (3.2% O.M.; pH 7.5; 11% field capacity moisture)	1.1 $kg \cdot ha^{-1}$	0–5 cm	71 ± 6 (Oct. 71–May 72)		
			31 ± 6 (Oct. 72–May 73)		
			14 ± 1 (May 72–Oct. 72)		
			14 ± 6 (May 73–Oct 73)		
			17 ± 4 (May 72–May 73)		
			7 ± 2 (May 72–Oct. 73)		
		5–10 cm	Negligible residues (<2%)		
Lincoln, Nebr.: Sharpsburg silty clay loam (2.8% O.M.; pH 5.8 in surface 15 cm of soil)			Chemical assays of soil samples taken in Sept. 1972 ($kg \cdot ha^{-1}$)		Burnside 1974
	0 $kg \cdot ha^{-1}$ (May)	0–20 cm	0	Residue values are averages of two soil depths (0–10 cm and 10–20 cm) not subject and subject to fall plowing (tandem discing to 12-cm depth plus harrowing). Weeds removed by hand.	
	0.56		0 (applied 1969)		
	1.12		0 (applied 1969)		
	2.24		0.01 (applied 1969)		
	0.56		0 (applied 1969–70)		
	1.12		0.01 (applied 1969–70)		
	2.24		0.06 (applied 1969–70)		
	0.56		0.02 (applied 1969–71)		
	1.12		0.06 (applied 1969–71)		
	2.24		0.15 (applied 1969–71)		

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues ($mg\,kg^{-1}$ except when noted)	Results and comment	Reference
	Residue values are averages of all application rates (i.e., untreated, 0.56 kg/ha, 1.12 kg/ha, and 2.24 kg/ha) and years (i.e., 1969, 1969–70, and 1969–71)	0–10 cm	0.04 (not plowed)	After long-term application (i.e., 1969–71), long-term carry-over increased on fall plowed plots (believed to reduce volatilization and photodecomposition losses, but other detoxification methods in moister soil might have increased).	
		10–20 cm	0.02 (not plowed)		
		0–10 cm	0.04 (fall plowed)		
		10–20 cm	0.03 (fall plowed)		
Indian Head, Sask.: Clay (2.9% O.M., pH 7.9, 85% of field capacity)	2.7 kg/ha (5% granular formulation) Application Oct. 1986	0–10 cm	NR	Incorporated to 9-cm depth by field cultivator and attached harrow; second incorporation to same depth in April 1987 and third incorporation 4 d later to 7.5 cm depth. Applications and incorporations of "aged" soils carried out in field and after 10 mo sampled, added to polystyrene cartons in lab. "Fresh" soils are those untreated from the field which are treated at "aged" application rate and incubated as above. Replicates were moistened to 85% of their field capacities and maintained with waterings every 2nd d. Dissipation described by first-order kinetics. Dissipation equations were calculated - Log^{10} (% herbicide at T days).	Smith, Aubin, and Derksen 1988

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
Loam (4.6% O.M., pH 7.4, 85% of field capacity)			NR	Clay - Aged 10 months: 2.009–0.00279 T, RMS = 0.96 - Fresh: 2.004–0.00325 T, RMS = 0.97 Loam - Aged 10 months: 1.979–0.00217 T, RMS = 0.94 - Fresh: 1.967–0.00271 T, RMS = 0.88	
Manitoba: - Red Deer River heavy clay (8.8% O.M., pH 6.5, field capacity 33.8%)	0.2–3.0 mg kg^{-1} (dry weight soil)		50% (+ 102 d)	Lab study involving incubation of treated soil and analysis for total trifluralin at various posttreatment time intervals. Each replicate was fertilized at the beginning of the study. Replicates were watered to field capacity and the loss compensated by weekly waterings.	Webster *et al.* 1978
- Gladstone clay loam (9.2% O.M., pH 7.8, field capacity 27.8%)			50% (+ 107 d)	First-order equation (i.e., $C/Co = e^{-kt}$) where C = concentration at time, Co = initial concentration, and k = reaction rate constant) provided closest expression of the degradation results. Loss rate constants were 0.0068, 0.0065, 0.0069, and 0.0087 d^{-1} for the Red Deer River heavy clay, Gladstone clay loam, Newdale loam, and Almasippi loamy sand, respectively.	
- Newdale loam (6.8% O.M., pH 7.4, field capacity 24.9%)			50% (+ 100 d)		
- Almasippi loamy sand (2.9% O.M., pH 7.7, field capacity 20.0%)					

Table C-1. Continued

Location/soil type (% organic matter; pH: moisture content)	Application rate (as % ai)	Soil depths measured	Residues ($mg \cdot kg^{-1}$ except when noted)		Results and comment	Reference
			Residues after 60 d			
			Autoclaved	Nonautoclaved		
Heavy soil (2.5% O.M., pH 8.5)	30 $mg \cdot kg^{-1}$ ^{3}H-labelled trifluralin	0–10 cm	4.2	11.2	Lab study in which 1 kg of each soil type placed in 15-cm diameter pots. Upper 10 cm moistened layers mixed with ^{3}H-labelled trifluralin to 30 $mg \cdot kg^{-1}$. Watered (every second day) and incubated away from sunlight for 2 mo. Autoclaving carried out prior to addition of ^{3}H-labelled trifluralin. The two depths measured (i.e., 0–10 cm and 10–15 cm) refer to soil extracted with solvent prior to combustion to remove nonextractable ^{3}H-residues.	Mostafa *et al.* 1982
		10–15 cm	1.0	3.3		
		Full depth (combustion bound)	18.0	9.8		
Light soil (1.5% O.M., pH 8.2)		0–10 cm	6.9	14.2		
		10–15 cm	2.1	3.4		
		Full depth (combustion bound)	17.0	10.0		
Sandy soil (0% O.M., pH 6.8)		0–10 cm	1.8	7.8		
		10–15 cm	0.6	2.5		
		Full depth (combustion bound)	26.5	13.5		
Graysville, Man.: - Almasippi very fine sandy loam (3.7% O.M., pH 7.6)	Year 1	0–5 cm	Means ± standard deviation		Fall treatments incorporated to 10-cm depth with tandem disc within 1 h of application and a 2nd incorporation in spring following seeding (flax). Spring treatments incorporated twice within 1 h of application. Plots fertilized prior to 2nd incorporation. Seeding date reported as May 18, 1978. Time posttreatment (i.e., (time of residue samples) given as the number of weeks after seeding.	Pchajek, Morrison, and Webster 1983
	0.84 $kg \cdot ha^{-1}$ (May 11, 1978)		0.6 (at application)			
			0.230 ± 0.026 (at seeding)			
			0.120 ± 0.007 (+ 6 wk)			
			0.067 ± 0.015 (+ 15 wk)			
	1.12 $kg \cdot ha^{-1}$ (Oct. 23, 1977)		0.8 (at application)			
			0.280 ± 0.028 (at seeding)			
			0.223 ± 0.024 (+ 6 wk)			
			0.109 ± 0.020 (+ 15 wk)			

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg·kg^{-1} except when noted)	Results and comment	Reference
	2.24 kg·ha^{-1} (Oct. 23, 1977)		1.6 (at application)		
			0.600 ± 0.043 (at seeding)		
			0.449 ± 0.049 (+ 6 wk)		
			0.155 ± 0.010 (+ 15 wk)		
	Year 2				
	0.84 kg·ha^{-1} (May 27, 1979)		0.6 (at application)	Seeding date June 5, 1979. Lower residue concentrations in year 2 considered to possibly reflect a much cooler and wetter spring in 1979 than in 1978.	
			0.140 ± 0.010 (at seeding)		
			0.066 ± 0.015 (+ 6 wk)		
			0.018 ± 0.006 (+ 18 wk)		
	1.12 kg·ha^{-1} (Nov. 1, 1978)		0.8 (at application)		
			0.215 ± 0.043 (at seeding)		
			0.143 ± 0.021 (+ 6 wk)		
			0.074 ± 0.005 (+ 18 wk)		
	2.24 kg·ha^{-1} (Oct. 23, 1977)		1.6 (at application)		
			0.542 ± 0.059 (at seeding)		
			0.386 ± 0.047 (+ 6 wk)		
			0.244 ± 0.010 (+ 18 wk)		
Charlottetown, P.E.I.: Charlottetown fine sandy loam (2.1% O.M., pH 6.1)	1.0 kg·ha^{-1} May 1978	0–10 cm	NR	Half-life (days ± standard error) of 164 ± 14.9.	Jensen, Ivany, and Kimball 1983
	Mid-June 1979		0.21 (+ 320 d)	Half-life of 148.0 ± 13.0.	

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
Kentville, N.S.: Somerset loamy sand (1.1% O.M., pH 5.2)	May 1978		NR	Half-life of 144.3 ± 31.3.	
	Mid-June 1979		0.27 (+ 320 d)	Half-life of 164.1 ± 15.4.	
				Fertilized and disced prior to application, herbicide incorporated with rototiller/disc harrow twice to 8–10 cm immediately following application and seeded (commercial peas) within 2 d. Suggested that a cooler climated and shorter growing season would prolong the persistence in eastern Canada.	
Lower Rio Grande Valley, Tex.: Hidalgo clay (1.7% O.M., pH 3.0, 31% water at field capacity)	1.1 kg·ha^{-1} (Nov. 1)	0–2.5 cm	50% (+ 3 wk)	Reported that data on percent activity with 0.6 kg·ha^{-1} application rate similar to that for 1.1 kg·ha^{-1} application rate. Application of herbicide onto disced and rotary-tilled soil, immediately incorporated into top 2.5 cm with rotary tiller and seeded (carrots) 1 d later. Total rainfall 14.7 cm in 12 wk following experiment initiation and soil described as warm and wet after first month.	Menges and Hubbard 1970
		0–5 cm	20% (+ 1 mo)		
			10% (+ 3 mo)		
			1% (+ 7 mo)		
			0% (13 mo)		

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depth measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
Beltsville, Md.: Hatboro silt loam (1.2% O.M.)	2.84 kg·ha^{-1} (August 8, 1975)	0–5 cm	29% (÷ 50 h) 50% (5–14 h)	Herbicide not incorporated but left on soil surface. Application of mixture included 3.55 kg·ha^{-1} heptachlor, 0.72 kg·ha^{-1} chlordane, and 5.11 kg·ha^{-1} dacthal.	Glotfelty *et al.* 1984
	2.80 kg/ha^{-1} (June 30, 1977)		13% (÷ 50 h) 50% (1–18 h)	Application of mixture included 1.10 kg·ha^{-1} lindane. Soil reported to be more compact than above due to previous rainfall. Herbicide loss from soil followed first-order kinetics.	
Salisbury, Md.: Norfolk sandy loam	2.50 kg·ha^{-1} (July 14)	0–5 cm	75% (÷ 50 h)	Application of mixture also included 3.75 kg·ha^{-1} heptachlor, 0.84 kg·ha^{-1} chlordane, and 0.62 kg·ha^{-1} lindane. For all experiments, mixtures were applied to bare soil and allowed to remain on the surface.	
Stoneville, Miss.: Bosket sandy loam (1.5% O.M., field capacity 0.33 bar moisture tension)	1.0 mg·kg^{-1}	NR (50-g samples)	50% (÷ 50 d range of 29–60 d)	Greenhouse study. Water emulsion of herbicide thoroughly mixed with soil and transferred to plastic pots 20 cm in diameter. Dissipation rates exhibit first-order kinetics (r values are 0.90 and 0.97 for the Bosket and Sharkey soils, respectively) ($P = 0.05$). Maintain at field capacity (i.e., 0.33 bar moisture tension). Dissipation rate has r value of 0.97.	Savage 1978
Sharkey clay (4.2% O.M., field capacity maintained as above)	1.0 mg·kg^{-1}		50% (÷ 91 d range of 45–124 d)		
	0.5 mg·kg^{-1}	NR (70-g samples)	50% (÷ 48 d)		

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
			50% (+ 30 d)	Pots without drainage holes and water added until free water remained (alternated with 20-d drying period for subsampling). Dissipation with r value of 0.90.	
	0.5 mg kg^{-1}	NR (70-g samples)	50% (+ 55 d)	Maintain at field capacity. Nonautoclaved soil (r value of 0.99).	
			50% (+ 56 d)	Autoclaved soil (r value of 0.97).	
			50% (+ 6 d)	Flooded as previously described. Nonautoclaved soil (r value of 0.98).	
			50% (+ 7 d)	Autoclaved soil (r value of 0.99).	
As, Norway: Loam (2.8% carbon, pH 5.6, 60°N and 80 m above sea level, 3.5 mo of 12 with soil temperatures above 10°C)	1 kg ha^{-1} (May 22) (Treflan used: 240 g ai l^{-1})	0–10 cm	0.3 (+ 17 wk)	Lowest value and no further decrease in next year. Maximum reached (4 wk after application (suggested to possibly be result of movement in the soil).	Solbakken *et al.* 1982
	5 kg ha^{-1}		1.1 (+ 51 wk)	Lowest value and no further decrease measured. Maximum levels as above.	
				Incorporation with rotary cultivator to approximately 15 cm immediately after spraying and sown with fodder rape.	

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
Holt, Norway: Loamy sand (6.2% carbon, pH 6.2, 70°N and 10 m above sea level. 2 mo of 12 with soil temperatures above 10°C)	1 kg ha^{-1} (June 15)		0.1 (+ 14 wk)	Lowest value and no further decrease observed.	
	5 kg ha^{-1}		0.4 (+ 66 wk)	Lowest value at experiment end.	
				Reapplications at both locations in the following year (May 15 for As and June 6 for Holt). The residue curves showed similar patterns to those of the first year.	
				Incorporation as for As.	
				Reported that the climate at Holt compared to As did not show extraordinary dry or wet periods or variations in temperature. However, soil appeared to have a greater influence on persistence than climate.	
				In the first year, May–Sept. mean temperatures and precipitation were respectively 12.9°C and 331 mm for As (both below normal) and 9.0°C and 264 mm for Holt (temperature above normal, but precipitation below normal).	
				Despite comparably lower precipitation and temperature and greater percent organic matter (apparently a reflection of %C), residue persistence was less at Holt than As. This trend contradicts the findings of other studies.	

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues ($mg\,kg^{-1}$ except where noted)	Results and comment	Reference
Regina, Sask.: Regina heavy clay (physical characteristics NR)	0.75 kg ha^{-1} May 77 (emulsifiable concentrate 399.1 kg m^{-3})	0–5 cm	% remaining (averages ± standard deviation) 30 ± 0% (+ 10 wk)	Herbicide incorporated into soil, but procedure NR.	Smith 1979
			19 ± 0% (+ 20 wk)		
	As above but mixed with triallate in a 1:2 ratio by weight		34 ± 1% (+ 10 wk)		
			31 ± 3% (+ 20 wk)		
	0.75 kg ha^{-1} May 78 (emulsifiable concentrate 399.1 kg m^{-3})		23 ± 2% (+ 10 wk)		
			16 ± 4% (+ 20 wk)		
	As above but mixed with triallate in a 1:2 ratio by weight		22 ± 1% (+ 10 wk)		
			12 ± 1% (+ 20 wk)		
White City sandy loam (physical characteristics NR)	0.75 kg ha^{-1} May 77 (emulsifiable concentrate 399.1 kg m^{-3})		31 ± 4% (+ 10 wk)		
			9 ± 2% (+ 20 wk)		
	As above but mixed with triallate in a 1:2 ratio by weight		26 ± 4% (+ 10 wk)		
			12 ± 1% (+ 20 wk)		
	0.75 kg ha^{-1} May 78 (emulsifiable concentrate 399.1 kg m^{-3})		28 ± 0% (+ 10 wk)		
			20 ± 1% (+ 20 wk)		
	As above but mixed with triallate in a 1:2 ratio by weight		37 ± 2% (+ 10 wk)		
			21 ± 3% (+ 20 wk)		

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
Nashville, Tenn.: Egam loam (1.41% O.M., pH 6.15, 1.5% slope and moderate to good permeability, 9% soil mixture - air dry soil weight basis at time of herbicide application)	0.84 kg ha^{-1} (emulsifiable concentrate 479.3 kg m^{-3})	0–30.5 cm	0.077 (June 28) 0.037 (July 14) 0.008 (Aug. 28) 0.007 (Sept. 28) 0.002 (Nov. 17)	Herbicide incorporated to 9.8-cm depth immediately following application and soybean planted June 20. The growing season was recorded to be wetter than normal, although August and September received sub-normal precipitation. Until August, daily temperatures were slightly below normal, but 3°F to 6°F above normal later in the season. Dissipation described by first-order kinetics with calculated half-life of 35.8 d, although this value is not supported by the raw data.	Duseja, Akuzuri, and Holmes 1980
	1.27 kg ha^{-1} (June 19)		0.109 (June 28) 0.037 (July 14) 0.033 (Aug. 28) 0.011 (Sept. 28) 0.004 (Nov. 17)		
	1.68 kg ha^{-1} (June 19)		0.134 (June 28) 0.02 (July 14) 0.027 (Aug. 28) 0.026 (Sept. 28) 0.009 (Nov. 17)	At an application rate of 1.68 kg ai ha^{-1}, 29.1% residue persisted after 9 d and 2.0% after 21 wk (based on theoretical concentration at time 0 (i.e., 0.46 mg kg^{-1}).	
Beason clay (1.69% O.M., pH 6.0, 0% slope with impeded drainage, soil moist 15% at time of application)	0.12 kg ha^{-1} (June 19)	0–30.5 cm	0.065 (June 28) 0.045 (July 14) 0.004 (Aug. 28) 0.003 (Sept. 28) 0 (Nov. 17)	Half-life of 25.7 d not supported by raw data. At an application rate of 1.68 kg ai ha^{-1}, 18.3% residue persisted after 9 d and 0.5% after 21 wk (based on theoretical concentration at time 0 (i.e., 0.42 mg kg^{-1}). Longevity basically the	

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
	1.68 kg ha^{-1} (June 19)		0.077 (June 28)	same in each soil although the clay soil was treated at higher rates of herbicide application.	
			0.020 (July 14)		
			0.009 (Aug. 28)		
			0.013 (Sept. 28)		
			0.002 (Nov. 17)		
	2.24 kg ai ha^{-1} (June 19)		0.188 (June 28)		
			0.042 (July 14)		
			0.039 (Aug. 28)		
			0.008 (Sept. 28)		
			0.003 (Nov. 17)		
Nashville, Tenn.: Egam loam (same plots used as in above study but physical characteristics of soil NR)	0.84 kg ai ha^{-1} (emulsifiable concentrate 479.3 kg cm^{-3}) (June 11)		0.272 (June 11)	Herbicide incorporated to 9.8-cm depth immediately following application and soybeans planted June 14. The growing season was recorded to be wetter than normal and the average daily temperatures were sub-normal. Degradation described by first-order kinetics with half-life of 27.1 d, although this value is not supported by the raw data.	Dureja, Akesson, and Holmes 1980
			0.092 (July 12)		
			0.014 (Aug. 23)		
			0.007 (Oct. 15)		
	1.27 kg ha^{-1} (June 11)		0.337 (June 11)		
			0.123 (July 10)		
			0.045 (Aug. 23)		
			0.019 (Oct. 15)		

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg·kg^{-1} except when noted)	Results and comment	Reference
	1.68 kg·ha^{-1} (June 11)		0.614 (June 11) 0.180 (July 12) 0.049 (Aug. 23) 0.018 (Oct. 15)	At an application rate of 1.68 kg·ha^{-1}, 29.3% residue persisted after 31 d and 2.9% after 18 wk (based on actual concentration at time 0, June 11)	
Beason clay (same as for Egan loam)	1.12 kg·ha^{-1} (June 11)		0.459 (June 11) 0.102 (July 12) 0.027 (Aug. 23) 0.009 (Oct. 15)	Half-life of 27.0 d not supported by raw data. At an application rate of 1.68 kg·ha^{-1}, 23.4% residue persisted after 31 d and 2.0% after 18 wk (based on actual concentration at time 0, June 11). Temperatures in 1978 study warmer and first 2 mo (June/July) received more rainfall (31.75 cm) than in 1980 study (22.1 cm). These climatic factors considered to contribute to higher residue levels in 1980 study.	
	1.68 kg·ha^{-1} (June 11)		0.602 (June 11) 0.141 (July 12) 0.031 (Aug. 23) 0.012 (Oct. 15)		
	2.24 kg·ha^{-1} (June 11)		0.914 (June 11) 0.176 (July 12) 0.044 (Aug. 23) 0.017 (Oct. 15)		
Bushland, Tex.: Pullman silty clay (1.6% O.M.)	NR (spring)	0–30.5 cm (on beds) 0–15.2 cm (in furrows)	36% (+ 6 wk) 14% (+ 3 mo)		Axt, Mathers, and Weiss 1969

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg·kg^{-1} except when noted)	Results and comment	Reference
Shafter, Calif. Panoche loam (<1% O.M.)	1.7 kg·ha^{-1} (each year for 6 years during last week of June or first week of July)	0-30 cm (average of 2 top 15-cm depth increments)	0.06 (year 1 at harvest) 0.10 (year 4 at harvest) 0.105 (year 5 at harvest) 0.075 (year 6 at harvest) -0.03 (1 yr after 1st application) -0.015 (1 yr after 2nd application) -0.01 (1 yr after 3rd application) -0.07 (1 yr after 4th application) -0.045 (1 yr after 5th application) -0.02 (1 yr after 6th application) -0.015 (1 yr after 3rd application) -0.015 (1 yr after 4th application) -0.045 (1 yr after 5th application) -not detectable (1 yr after 6th yr application)	Each year of study cotton plants shredded and returned to soil by disking. Applied as direct spray to base of cotton plants at time of last cultivation and incorporated same day with two passes of a rolling cultivator. Two furrow irrigations per year (first within a few days of application). Chemical analysis at harvest of cotton. From 30 to 120 cm depth, residue was not detectable (limit of detection was about 0.02 mg·kg^{-1} for year 4 for each 15-cm depth increment below 30 cm and for year 1 where analyses not obtained below 30 cm (this refers to analysis at harvest). Harvest approximately 6 mo after application. For 0.8 kg·ha^{-1} application, samples were not taken for 1 yr after each of first and second yr application.	Miller *et al.* 1978
	0.8 kg·ha^{-1}				
Nashville, Tenn.: Etowah silt loam (1.79% O.M., pH 6.4, 64% field capacity)		15 cm (used for lab study)		Lab study (soil with stated properties removed from field, dried, ground, passed through sieve and "spiked" with herbicide); 40 g incubated in styrofoam	Duseja 1982

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
pH 5.3, 50% soil moisture	0.5 mg kg^{-1}	10 g of each 40 g per replicate		cups and moisture replenished every 24 h. Soil moistures maintained in the lab as 50% and 100% of the field capacity. Residue values followed by same letter within same time interval are not significantly different at 1% level according to Duncan multiple range test.	
22.8°C			87.3% (+ 7 d)		
35.0°C			69.7% (+ 7 d)		
			34.7% (+ 17 d)		
100% soil moisture					
35.0°C			34.8% (+ 7 d)		
			25.5% (+ 17 d)		
pH 6.4, 50% soil moisture					
22.8°C			80.0% (+ 7 d)		
35.0°C			57.6% (+ 7 d)		
			24.2% (+ 17 d)		
100% soil moisture					
35.0°C			24.2% (+ 7 d)		
			20.0% (+ 17 d)		
pH 7.5, 50% soil moisture					
22.8°C			89.7% (+ 7 d)		
35.0°C			74.3% (+ 7 d)		
			37.9% (+ 17 d)		
100% soil moisture					
35.0°C			48.5% (+ 7 d)		
			30.3% (+ 17 d)		

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
Kentville, N.S.: Berwick loamy sand (4.2% O.M., pH 5.2)	1.0 kg ha^{-1} (Nov. 26, 1977)	0–10 cm (20-g sub-samples)	0.42 (Nov. 26/77)	Incorporated to approximately 10 cm immediately after application with a rototiller. Following spring application, entire plot tilled and planted to peas; No difference ($P = 0.5\%$) between soils in terms of overwinter losses (i.e. from Nov. 26/77 to Apr. 16/78). Reported that "the relatively long period of high soil moisture following the spring thaw may favour" the measured overwinter losses. Half-life of 126 d following the spring application on Berwick loamy sand.	Jensen and Kimball 1980
			0.26 (Apr. 16/78)		
			0.18 (May 16/78)		
			0.16 (June 20/78)		
			0.14 (Aug. 29/78)		
			0.12 (Nov. 23/78)		
	0.75 kg ha^{-1} (May 16, 1978)		0.34 (May 16/78)		
			0.24 (June 20/78)		
			0.19 (July 20/78)		
			0.20 (Aug. 29/78)		
			0.15 (Nov. 23/78)		
Somerset loamy sand (1.3% O.M., pH 5.3)	1.0 kg ha^{-1} (Nov. 26, 1977)		0.36 (Nov. 26/77)		
			0.22 (Apr. 16/78)		
			0.23 (May 16/78)		
			0.17 (June 20/78)		
			0.17 (Aug. 29/78)		
			0.17 (Nov. 23/78)		
	0.75 kg ha^{-1} (May 16, 1978)		0.38 (May 16/78)	Over the 190-d spring test period, the herbicide level did not dissipate to 50% in the Somerset loamy sand despite organic matter content lower than the Berwick loamy sand.	
			0.40 (June 20/78)		
			0.30 (July 20/78)		
			0.32 (Aug. 29/78)		
			0.28 (Nov. 23/78)		

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
Buchland, Tex.: Sand (1.3% O.M., pH 5.3)	0.8 kg ha^{-1} (2 wk before late May/81 - preplant incorporated)	0–7.5 cm	0.15 (Aug. 1/81) 0.04 (Oct. 15/81) 0.08 (Feb. 18/82)	Sprayed broadcast and incorporated with a rolling cultivator approximately 2 wk before planting soybeans in late May (preplant incorporated). Sprayed broadcast at layby on soybeans and incorporated as above in mid-July. Received preplant irrigation, rainfall and four additional irrigations during the summer (total of 50 cm irrigated water each yr). Experiment repeated the following yr (i.e., May/81 and May/82); 24.6 cm rainfall in the 2nd week of Aug. 1981, shortly after irrigation and flooded the level borders for 3 d (believed that the anaerobic conditions probably caused rapid breakdown of the herbicide). The residue decrease was not as great in 1982 experiment where flooding did not occur.	Warner, Winter, and Weise 1987
	0.8 kg ha^{-1} (mid-July-layby)		0.22 (Aug. 1/81) 0.10 (Oct. 15/81) 0.06 (Feb. 18/82)		
	0.8 kg ha^{-1} (2 wk before late May/82 - preplant incorporated)		0.36 (Aug. 20/82) 0.22 (Oct. 15/82) 0.14 (Mar. 15/83)		
	0.8 kg ha^{-1} (mid-July-layby)		0.40 (Aug. 20/82) 0.36 (Oct. 15/82) 0.32 (Mar. 15/83)		
Woodslee, Ont.: Brookston clay loam (3.6% O.M., pH 6.6)	1.0 kg ai ha^{-1} May 23, 1979	0–10 cm	50% (= 116–173 d)	Applied with metribuzin (0.5 kg ai ha^{-1}) preplant incorporated (to 10 cm with a disc in 2 directions) to soybeans: 1st order rate of loss identified t 1/2 (half-life in days)= ln 2/K: k(regression coefficient) = 0.005 ± 0.001 and R^2(coefficient of determination) = 0.70 for Brookston clay loam, 1979; k = 0.005 ± 0.003 and R^2 = 0.39 for 1980.	Gaynor 1985
	June 21, 1980		35% (= 110 d)		

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
Fox sandy loam Brookston clay loam (3.6% O.M., pH 6.6)	2.0 kg ai ha^{-1} (April 17, 1980)			Applied with metribuzin (0.75 kg ai ha^{-1}) and incorporated to 10 cm in 2 directions with a disc;	
		0–10 cm	50% (+ 63-77 d)	$k = 0.010 \pm 0.001$ and $R^2 = 0.92$;	
		10–20 cm	50% (+ 69-116 d)	$k = 0.008 \pm 0.002$ and $R^2 = 0.78$.	
		Full depth	35% (+ 90 d)	These depths were sampled, but the tabulated data for each were not specifically designated to them, therefore, the residue-depth matchups for the 2.0 kg ai ha^{-1} are assumed.	
				Gaynor (1985) attempted to assess soil persistence in southwestern Ontario soils as compared to those of the prairies and the maritimes (average temperatures are 2°C–4°C higher in southwestern Ontario, and this area receives 50% more rainfall than south Saskatchewan, but 21%–25% less than the Maritimes), but suggests that soil type does not have an effect on persistence in this case (the data do not support his conclusion) and may be related to the low organic matter content (less than 4%).	

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
Regina, Sask. Regina heavy clay (physical characteristics NR)	1 kg ha^{-1} May 4, 1977	0–5 cm	16% (+ 10 wk)	Incorporated to 5-cm depth with a small fork. Less than 2% of the applied herbicide was found in the 5–10 cm soil depth; over 64% lost in first 10 wk; Minimal overwintering losses. Slower losses from the soils in 1979 attributed to later application date and drier conditions (10.8 cm precipitation compared to 25.4 cm and 27.8 cm for 1977 and 1978, respectively, from time of application until freeze-up), which would reduce volatilization (considered to be the most important means of dissipation).	Hayden and Smith 1980
			9% (+ 20 wk)		
			6% (+ 52 wk)		
	May 4, 1978		27% (+ 10 wk)		
			16% (+ 20 wk)		
			10% (+ 55 wk)		
	May 31, 1979		36% (+ 10 wk)		
			29% (+ 20 wk)		
			20% (+ 48 wk)		
White City sandy loam (physical characteristics NR)	May 4, 1977		20% (+ 10 wk)		
			7% (+ 20 wk)		
			7% (+ 52 wk)		
	May 4, 1978		34% (+ 10 wk)		
			19% (+ 20 wk)		
			13% (+ 55 wk)		
	May 31, 1979		30% (+ 10 wk)		
			25% (+ 20 wk)		
			15% (+ 48 wk)		
Manitoba (physical characteristics NR)	NR (May application)	NR	17%–26% (+ 50 wk)	Review paper	Smith 1983

Table C-1. Continued

Location/soil type (% organic matter: pH, moisture content)	Application rate (as % ai)	Soil depths measured	Residues ($mg\,kg^{-1}$ except when noted)	Results and comment	Reference
Saskatchewan (physical characteristics NR)	NR (Spring application)	NR	8%–53% (following October)		
	NR (Fall application)	NR	17%–71% (following spring)		
Nova Scotia (physical characteristics NR)	NR (May application)		44%–74% (+ 28 wk)		
	NR (Fall application)		60% (following spring)		
Regina, Sask.: Regina heavy clay (4.2% O.M., pH 7.3, field capacity 40%)	1.1 $kg\,ha^{-1}$ (May 1972, 1973 and 1974)	0–5 cm	12% ± 4%, 8% ± 2%, 11% ± 3% (avg. 10% ± 2%) + 5 mo	Incorporation to 5 cm depth for 2 min. with small fork. <2% of the herbicide applied was recovered from the 5–10 cm soil depths.	Smith and Hayden 1976
	May 1972, 1973		16% ±5%, 5% ± 0% + 12 mo		
	May 1972, 1973		3% ± 0%, 3% ± 0% + 17 mo		
Jamroon, Sask.: Asquith sandy loam (3.2% O.M., pH 6.7, field capacity 12%)	1.1 $kg\,ha^{-1}$ (May 1972, 1973 and 1974)		14% ± 1%, 14% ± 6%, 11% ± 4% (avg. 13% ± 2%) + 5 mo	Incorporation to 5-cm depth for 2 min. with small fork. <2% of the herbicide applied was recovered from the 5–10 cm soil depths.	
	May 1972, 1973		17% ± 4%, 9% ± 3% + 12 mo		
			7% ± 2%, 2% ± 0% + 17 mo		
Melfort, Sask.: Melfort silty loam (11.7% O.M., pH 6.2, field capacity 35%)	1.1 $kg\,ha^{-1}$		31% ± 7%, 24% ± 6%, 15% ±4% (avg. 23% ± 8%) + 5 mo	Incorporation to 5-cm depth for 2 min with small fork. <2% of the herbicide applied was recovered from the 5–10 cm soil depths.	
			35% ± 3%, 19% ± 3% + 12 mo		
			16% ± 3%, 14% ± 2% + 17 mo		

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comment	Reference
Saskatchewan: Sandy loam (soil conditions NR)	0.75 kg ha^{-1} 2nd wk May 1979	0-5 cm	33% ± 3% ÷ 22 wk	Incorporation to 5-cm depth with a small fork. Plots left fallow and hand weeded. No residues found below 5-cm depth. Losses of trifluralin are significantly affected by the addition of chloramben.	Smith and Hayden 1982a
	2nd wk May 1980		39% ± 4% ÷ 22 wk		
	2nd wk May 1981		13% ± 2% ÷ 22 wk		
	0.75 kg ha^{-1} + 2.0 kg ha^{-1} chloramben 2nd wk May 1979		25% ± 2% ÷ 22 wk		
	2nd wk May 1980		40% ± 2% ÷ 22 wk		
	2nd wk May 1981		13% ± 2% ÷ 22 wk		
Heavy clay (soil conditions NR)	0.75 kg ha^{-1} 2nd wk May 1979		40% ± 4% ÷ 22 wk		
	2nd wk May 1980		53% ± 3% ÷ 22 wk		
	2nd wk May 1981		16% ± 11% ÷ 22 wk		
	0.75 kg ha^{-1} + 2.0 kg ha^{-1} chloramben 2nd wk May 1979		47% ± 5% ÷ 22 wk		
	2nd wk May 1980		45% ± 6% ÷ 22 wk		
	2nd wk May 1981		15% ± 3% ÷ 22 wk		
Stoneville, Miss.: Bosket silt loam (physical characteristics NR)	0.84 kg ha^{-1}	Full depth (10 cm)	0.10 ÷ 8 wk (1.3 cm incorporation depth)	Half-gallon cartons filled with 2000 g of soil; appropriate depth of soil removed and thoroughly mixed with trifluralin to achieve application rate of 0.84 kg ha^{-1}. Planted with cotton and morningglory for 3 wk, chipped and planted with corn. Grown in greenhouse.	Savage and Barrentine 1969
		Full depth (10 cm)	0.16 ÷ 8 wk (3.8 cm incorporation depth)		
		Full depth (10 cm)	0.22 ÷ 8 wk (7.6 cm incorporation depth)		

Table C-1. Continued

Location/soil type (% organic matter; pH; moisture content)	Application rate (as % ai)	Soil depths measured	Residues (mg kg^{-1} except when noted)	Results and comments	Reference
Moisture maintained at field capacity (NR)	1.12 kg ha^{-1}	Full depth (10 cm)	0.19 + 8 wk (1.3 cm incorporation depth)	As above but no plants grown.	
		Full depth (10 cm)	0.25 + 8 wk (3.8 cm incorporation depth)		
		Full depth (10 cm)	0.30 + 8 wk (7.6 cm incorporation depth)		
As above; but soil moisture NR	0.84 kg ha^{-1} May 12, 1967		July 28, 1967 (+ 12 wk)	Herbicide applied in field to a 51-cm band on top of preformed rows. Incorporation treatment includes surface application, incorporation to 2.5–5 cm using a double lawn mower reel incorporator and incorporation to 7.6–10 cm with a power cultivator.	
		0–10 cm	0.015 (surface application)		
		0–10 cm	0.18 (2.5–5 cm incorporation depth)		
		0–10 cm	0.305 (7.6–10 cm incorporation depth)		
			January 22, 1968 (+ 40 wk)		
		0–10 cm	0.015 (surface application)		
		0–10 cm	0.055 (2.5–5 cm incorporation depth)		
		0–10 cm	0.020 (7.6–10 cm incorporation depth)		
	4.48 kg ha^{-1} May 12, 1967		January 28, 1967 (+ 12 wk)		
		0–10 cm	0.06 (surface application)		
		0–10 cm	0.735 (2.5–5 cm incorporation depth)		
		0–10 cm	1.03 (7.6–10 cm incorporation depth)		

Table C-1. Continued

Location/soil type (% organic matter, pH, moisture content)	Application rate (or % ai)	Soil depth measured	Residues (mg kg^{-1} except when noted)	Results and comments	Reference
			December 19, 1967 (+ 36 wk)		
		0-10 cm	0.02 (surface application)		
		0-10 cm	0.22 (2.5-5 cm incorporation depth)		
		0-10 cm	0.415 (7.6-10 cm incorporation depth)		
Regina, Sask.: Heavy clay (physical characteristics NR)	1.25 kg ha^{-1} 1st wk September	0-5 cm	37% 2nd wk of following May (average for 5 yr)	Incorporation to 5-cm depth with a small fork. With exception of one application years, all data within other application years not significantly different at the 0.01 level (Duncan's multiple range test) whether different soil type or different month of application. This suggests that the difference between the average values is also quite small. Less than 2% of the applied herbicide detected in the 5-10 cm depths.	Smith and Hayden 1982b
	1st wk October	0-5 cm	38.6% 2nd wk of following May (average for 5 yr)		
	1st wk November	0-5 cm	38.6% 2nd wk of following May (average for 5 yr)		
Sandy loam (physical characteristics NR)	1st wk September	0-5 cm	31.2% 2nd wk of following May (average for 6 yr)		
	1st wk October	0-5 cm	33.3% 2nd wk of following May (average for 6 yr)		
	1st wk November	0-5 cm	34.3% 2nd wk of following May (average for 6 yr)		
Haifa, Israel 1976 Sieves to air soil (2.5% O.M., 38% of field capacity, pH NR)	4.0 mg kg	NR	2.4 mg kg^{-1} (10°C)	Soil sieved, mixed with trifluralin, and 1 kg placed in a double polyethylene bag. Incubated in the dark at 10°C, 20°C, 30°C, or 40°C. Water lost by evaporation was replaced to initial moisture level. Incubated for 2 mo.	Haroun, Helm, and Blumenfeld
	4.0 mg kg	NR	1.0 mg kg^{-1} (20°C)		
	4.0 mg kg	NR	0.8 mg kg^{-1} (30°C)		
	4.0 mg kg	NR	0.5 mg kg^{-1} (40°C)		
	8.0 mg kg	NR	6.4 mg kg^{-1} (10°C)		
	8.0 mg kg	NR	3.5 mg kg^{-1} (20°C)		
	8.0 mg kg	NR	1.2 mg kg^{-1} (30°C)		
	8.0 mg kg	NR	0.8 mg kg^{-1} (40°C)		

Appendix D

Observed or Calculated Trifluralin Bioconcentration Factors in Aquatic Biota

Table D-1. Observed or Calculated Trifluralin Bioconcentration Factors in Aquatic Biota

Species/ tissue	Exposure medium (single, continuous)	Formulation (% ai)	BCF	Treatment duration	Comments	Reference
Fish (*Gambusia affinis*)	Static exposure to water containing 0.2 to 0.9 µg L^{-1} over 30-d period	Trifluralin (>97%)	1000 3140 750 5750 2630	1 d 3 d 7 d 15 d 30 d	Static microcosm study with ^{14}C-trifluralin introduced adsorbed to soil. Continuous-flow microcosms received trifluralin dissolved in water via acetone carrier. BCF based on ratios of ^{14}C in water and tissue. No discrimination of metabolites.	Yoshimi, Inouye, and Walker 1980
	Static exposure to water containing 3.4 to 9.1 µg L^{-1} over 30-d period	Trifluralin (>97%)	320 1080 380 500 300	1 d 3 d 7 d 15 d 30 d		
	Static exposure to water containing 36.9 to 160.1 µg L^{-1} over 30-d period	Trifluralin (>97%)	690 1150 350 80 70	1 d 3 d 7 d 15 d 30 d		
	Continuous-flow exposure to water containing 0.1 to 0.8 µg L^{-1} over 30-d period	Trifluralin (>97%)	11000 3000 6000 1800 3250	1 d 3 d 7 d 15 d 30 d		
	Continuous-flow exposure to water containing 0.5 to 2.6 µg L^{-1} over 30-d period	Trifluralin (>97%)	2200 2670 5710 2080 5080	1 d 3 d 7 d 15 d 30 d		
	Continuous-flow exposure to water containing 9.3 to 29.8 µg L^{-1} over 30-d period	Trifluralin (>97%)	1190 1910 3960 4050 3810	1 d 3 d 7 d 15 d 30 d		
Snail (*Helisoma sp.*)	Static exposure to water containing 0.2 to 0.9 µg L^{-1} over 30-d period	Trifluralin (>97%)	1000 140	1 d 3 d	Static microcosm study with ^{14}C-trifluralin introduced adsorbed to soil. Continuous-flow microcosms received trifluralin dissolved in water via acetone carrier. BCF based on ratios of ^{14}C in water and tissue. No discrimination of metabolites.	
	Static exposure to water containing 3.4 to 9.1 µg L^{-1} over 30-d period	Trifluralin (>97%)	150 200 40 10	1 d 3 d 15 d 30 d		
	Static exposure to water containing 36.9 to 160.1 µg L^{-1} over 30-d period	Trifluralin (>97%)	150 140 40 20 20	1 d 3 d 7 d 15 d 30 d		

BCF = concentration in organism or bioaccumulation in medium (water or diet).
NR = not reported

Table D-L. Continued

Species/ tissue	Exposure medium (single, continuous)	Formulation (% ai)	BCF	Treatment duration	Comments	Reference
	Continuous-flow exposure to water containing 0.1 to 0.8 µg L^{-1} over 30-d period	Trifluralin (>97%)	2000 500 1000 600 130	1 d 3 d 7 d 15 d 30 d		
	Continuous-flow exposure to water containing 9.3 to 29.8 µg L^{-1} over 30-d period	Trifluralin (>97%)	270 1110 1590 1090 870	1 d 3 d 7 d 15 d 30 d		
Green, filamentous alga (*Oedogonium cardiacum*)	Static exposure to water containing 0.2 to 0.9 µg L^{-1} over 30-d period	Trifluralin (>97%)	1000 290 250	1 d 3 d 15 d	Static microcosm study with ^{14}C-trifluralin introduced adsorbed to soil. Continuous-flow microcosms received trifluralin dissolved in water via acetone carrier. BCF based on ratios of ^{14}C in water and tissue. No discrimination of metabolites.	
	Static exposure to water containing 3.4 to 9.1 µg L^{-1} over 30-d period	Trifluralin (>97%)	500 260 240 210 250	1 d 3 d 7 d 15 d 30 d		
	Static exposure to water containing 36.9 to 160.1 µg L^{-1} over 30-d period	Trifluralin (>97%)	1030 160 280 210 220	1 d 3 d 7 d 15 d 30 d		
	Continuous-flow exposure to water containing 0.1 to 0.8 µg L^{-1} over 30-d period	Trifluralin (>97%)	4500 20000 2600 1660	3 d 7 d 15 d 30 d		
	Continuous-flow exposure to water containing 0.5 to 2.6 µg L^{-1} over 30-d period	Trifluralin (>97%)	2550 1560 23660 6270 1240	1 d 3 d 7 d 15 d 30 d		
	Continuous-flow exposure to water containing 9.3 to 29.8 µg L^{-1} over 30-d period	Trifluralin (>97%)	280 2560 6930 3770 4750	1 d 3 d 7 d 15 d 30 d		
Water flea (*Daphnia magna*)	Static exposure to water containing 0.2 to 0.9 µg L^{-1} over 30-d period	Trifluralin (>97%)	1000 160 1250 110	1 d 3 d 7 d 30 d	Static microcosm study with ^{14}C-trifluralin introduced adsorbed to soil. BCF based on ratios of ^{14}C in water and tissue. No discrimination of metabolites.	
	Static exposure to water containing 3.4 to 9.1 µg L^{-1} over 30-d period	Trifluralin (>97%)	560 1060 280 20 40	1 d 3 d 7 d 15 d 30 d		

Table D-1. Continued

Species/ tissue	Exposure medium (single, continuous)	Formulation (% ai)	BCF	Treatment duration	Comments	Reference
	Static exposure to water containing 36.9 to 160.1 µg L^{-1} over 30-d period	Trifluralin (>97%)	510 630 250 40 30	1 d 3 d 7 d 15 d 30 d		
Fathead minnow (*Pimephales promelas*)	Static exposure to water containing 20 µg L^{-1} for 40 h	Reagent grade (NR)	2361	40 h		Spacie and Hamelink 1979
Fish (various species)	River water (concentration of 0.0018 mg L^{-1})	NR	1800–6000	NR	It was reported that trifluralin is accumulated by direct uptake from water; fish residues were proportional to concentration in the river water.	Thierron-Richards and Williamson 1987
Fish	Water (concentration of 1.8×10^{-4} mg L^{-1})	NR	7200	NR	Water concentration refers to that for trifluralin plus metabolites.	Williamson 1984
Snails	Water (concentration of 1.8×10^{-4} mg L^{-1})	NR	157000	NR	Water concentration refers to that for trifluralin plus metabolites.	
Algae (*Oedogonium cardiacum*)	Static water microcosms	NR	276	33 d	4 L aquatic microcosms containing ^{14}C-trifluralin adsorbed to soil. *Daphnia*, snails, algae, and "old aquarium water" added immediately. After 30 d, *Daphnia* removed and two fish added. All organisms harvested 3 d later.	Kearney, Isensee and Kontson 1977
Snails (*Helisoma* sp.)	Static water microcosms	NR	630	33 d	4 L aquatic microcosms containing ^{14}C-trifluralin adsorbed to soil. *Daphnia*, snails, algae, and "old aquarium water" added immediately. After 30 d, *Daphnia* removed and two fish added. All organisms harvested 3 d later.	
Cladocera (*Daphnia magna*)	Static water microcosms	NR	92	30 d	4 L aquatic microcosms containing ^{14}C-trifluralin adsorbed to soil. *Daphnia*, snails, algae, and "old aquarium water" added immediately. After 30 d, *Daphnia* removed and two fish added. All organisms harvested 3 d later.	

Table D-1. Continued

Species/ tissue	Exposure medium (single, continuous)	Formulation (% ai)	BCF	Treatment duration	Comments	Reference
Mosquitofish (*Gambusia affinis*)	Static water microcosms	NR	33	3 d	4 L aquatic microcosms containing ^{14}C-trifluralin adsorbed to soil. *Daphnia*, snails, algae, and "old aquarium water" added immediately. After 30 d, *Daphnia* removed and two fish added. All organisms harvested 3 d later.	
Fathead minnow (*Pimephales promelas*) (eviscerated carcass)	Continuous exposure to water containing 5.1 µg L^{-1} (mean conc.)	Technical (99)	961	425 d		Mauck *et al.* 1978
	Continuous exposure to water containing 1.9 µg L^{-1} (mean conc.)	Technical (99)	1333	425 d		
	Continuous exposure to water containing 1.5 µg L^{-1} (mean conc.)	Technical (99)	889	425 d		

Appendix E

Acute Toxicity of Trifluralin to Aquatic Organisms

Table B-1. Acute Toxicity of Trifluralin to Aquatic Organisms

Species	Test conditions*	Temperature (°C)	pH	Hardness (mg $CaCO_3$ L^{-1})	Formulation (% ai)	LC_{50}/EC_{50} (mg L^{-1}) 24 h (confidence interval)	48 h	96 h	Reference
VERTEBRATES									
Salmo gairdneri (Rainbow trout)	S, M	2.0	7.4	44	Technical (95.9)	560 (471–666)	330 (281–387)		Mayer and Ellersieck 1986
	S, M	7.0	7.4	44	Technical (95.9)	250 (218–287)	120 (98–147)		
	S, M	12.0	7.4	44	Technical (95.9)	167 (149–188)	92		
	S, M	18.0	7.4	44	Technical (95.9)	100 (79–127)	<34		
	S, M	12.0	7.4	44	Emul. conc. (48)	13.5 (11.3–16.2)	10 (7.2–14)		
	S, M	12.0	7.4	44	Emul. conc. (48)	210 (161–275)	76 (52–111)		
	S, M	12.0	7.4	44	Emul. conc. (48)	120 (86–167.5)			
	S, M	12.0	7.4	44	Emul. conc. (48)	135 (92.5–198.8)	95 (71.5–134.1)		
	S, M	12.0	7.4	44	Emul. conc. (48)	95 (66–144)	28 (18–42)		
	S, M	12.0	7.4	44	Emul. conc. (48)	96 (67–138)	50 (34–74)		
	S, M	12.0	7.4	44	Emul. conc. (48)	78 (51–118)	41 (26–62)		
	S, M	12.0	7.4	320	Emul. conc. (48)	92 (63–134)	43 (28–66)		
	S, M	12.0	7.5	44	Emul. conc. (48)	86 (58–126)	42 (27–65)		
	S, M	12.0	6.5	44	Emul. conc. (48)	56 (38–83)	33 (24–46)		
	S, M	12.0	8.5	44	Emul. conc. (48)	43 (25–73)	25 (15–41)		

Note: Trifluralin acute toxicity data reported by Johnson and Finley (1980) was also reported in Mayer and Ellersieck (1986).

*Test conditions: S = static
F = flow-measured
M = measured
V = unmeasured

NR = not reported

Table 5-1. Continued

Species	Test conditions*	Temperature (°C)	pH	Hardness (mg $CaCO_3$ L^{-1})	Formulation (% ai)	LC_{50}/EC_{50} (mg L^{-1}) 24 h (confidence interval)	48 h (confidence interval)	96 h (confidence interval)	Reference
	S, M	7.0	7.8	44	Emul. conc. (40%)		100 (58–172)		
	S, M	12.0	7.8	44	Emul. conc. (40%)		60 (37–98)		
	S, M	17.0	7.8	44	Emul. conc. (40%)		22 (16–30)		
	S, M	12.0	7.8	42	Emul. conc. (40%)	53 (39–72)	51 (36–73)		
(fingerling)	S, M	12.0	7.8	44	Emul. conc. (40%)	130 (99–180)	86 (61–120)		
(swimup fry)	S, M	12.0	7.8	44	Emul. conc. (40%)	>1000	83 (53–130)		
	S, M	12.0	7.8	40	Emul. conc. (40%)	>1000	140 (89–240)		
	S, M	12.0	7.8	40	Emul. conc. (40%)	370 (270–510)	170 (100–280)		
	S, M	12.0	7.8	42	Emul. conc. (40%)	430 (310–540)	160 (96–270)		
(yolk-sac fry)	S, M	12.0	7.8	44	Emul. conc. (40%)	>1000	1600 (1200–2100)		
Carassius auratus (Goldfish)	S, M	18.0	7.8	44	Emul. conc. (40%)	700 (458–1068)	145 (108–195)		Mayer and Ellersieck 1986
Pimephales promelas (Fathead minnow)	S, M	18.0	7.8	44	Technical (95.9)	350 (265–450)	160 (116–220)		Mayer and Ellersieck 1986
	S, M	18.0	7.8	44	Technical (95.9)	205 (167–251)	124 (95–162)		
	S, M	18.0	7.8	44	Technical (95.9)	148 (121–182)	105 (83–134)		
Ictalurus punctatus (Channel catfish)	S, M	18.0	7.8	44	Technical (95.9)	500 (424–589)			Mayer and Ellersieck 1986
	S, M	18.0	7.8	44	Technical (95.9)	650 (436–969)	440 (361–536)		
	S, M	24.0	7.8	44	Technical (95.9)	400 (198–809)	210 (135–375)		
	S, M	22.0	7.8	44	Technical (95.9)	6400 (2400–7860)	2500 (1420–3410)		

Table 3-1. Continued

Species	Test conditions[a]	Temperature (°C)	pH	Hardness (mg $CaCO_3$ L^{-1})	Formulation (% ai)	LC_{50}/EC_{50} (mg L^{-1}) 24 h	48 h (confidence intervals)	96 h	Reference
[illegible]	S, M	22.0	7.6	40	Technical (95.9)		330 (215–511)		
[illegible]	S, M	22.0	7.6	40	Technical (95.9)		680 (520–830)		
[illegible]	S, M	24	7.6	44	Technical (95.9)	22.5 (19.9–25.2)	18.5 (16.8–19.9)		Mayer and Ellersieck 1986
	S, M	7	7.6	44	Technical (95.9)	1300 (1000–1700)	250 (240–330)		
	S, M	12	7.6	44	Technical (95.9)	530 (440–640)	210 (170–250)		
	S, M	18	7.6	44	Technical (95.9)	360 (300–430)	135 (120–160)		
	S, M	24	7.6	44	Technical (95.9)	120 (100–140)	67 (60–55)		
	S, M	29	7.6	44	Technical (95.9)	10 (8–13)	8.4 (6.5–11)		
	S, M	22	7.6	44	Technical (95.9)	77 (62.7–94.6)	60 (45.7–71.9)		
	S, M	22	7.6	44	Technical (95.9)	64 (54–87)	58 (47–70)		
	S, M	12.0	7.6	44	Technical (95.9)	>5000	400 (300–560)		
	S, M	17.0	7.6	44	Technical (95.9)		240 (170–330)		
	S, M	22.0	7.6	44	Technical (95.9)	400 (340–430)	190 (130–280)		
	S, M	22.0	6.5	40	Technical (95.9)		100 (64–160)		
	S, M	22.0	7.5	40	Technical (95.9)		260 (190–390)		
	S, M	22.0	8.5	40	Technical (95.9)		120 (87–165)		
	S, M	22.0	7.5	40	Technical (95.9)	440 (295–561)	140 (45–256)		

Table E-1. Continued

Species	Test conditions[a]	Temperature (°C)	pH	Hardness (mg $CaCO_3$ L^{-1})	Formulation (% ai)	LC_{50} (EC_{50}, mg L^{-1}) 24 h (confidence intervals)	48 h (confidence intervals)	96 h (confidence intervals)	Reference
	S, M	22.0	7.6	120	Technical (95.9)	400 (302–530)	70 (47–104)		
Micropterus salmoides (Largemouth bass)	S, M	18.0	7.6	272	Technical (95.9)	120 (92–157)		70 (65–87)	Mayer and Ellersieck 1986
Menidia beryllina (Inland silverside)	S, M	18.0	7.6	44	Technical (95.9)	190 (125–286)			Mayer and Ellersieck 1986
Salvelinus fontinalis (Brook trout)	S, M	15.0	7.6	44	Technical (95.9)	280 (154–284)		115 (82–161)	Mayer and Ellersieck 1986
	S, M	15.0	7.6	44	Technical (95.9)	320 (188–330)		110 (66–183)	
Gambusia affinis (Mosquito fish)	S, U	23–25	NR	NR	Toxaphene (NR)	28 (26–35)		12 (11–13)	Nagel and Leong 1983
Gambusia affinis (Mosquito fish) ("susceptible")	S, U	25	NR	NR	NR	2.80			Schuster and Chamberlain 1974
("resistant")	S, U	25	NR	NR	NR	4.80			
Lepomis macrochirus (Bluegill)	S, U	23–25	NR	90	Toxiflow (4%)			58.2 (51–13)	Parks and Worth 1965
Pimephales promelas (Fathead minnow)	S, U	23–25	NR	90	Toxiflow (4%)			97.9–151.0	Parks and Worth 1965
Carassius auratus (Goldfish)	S, U	23–25	NR	90	Toxiflow (4%)			580	Parks and Worth 1965
Lepomis macrochirus (Bluegill)	S, U	20	NR	NR	NR		0.019		Cope 1966
Salmo gairdneri (Rainbow trout)	S, U	13	NR	NR	NR		0.011		Cope 1966
Rasbora heteromorpha (Harlequin fish)	F, U	20	7.2	250	Toxiflow (4%)	0.9			Alabaster 1969
Ictalurus punctatus (Channel catfish) (fingerling)	S, U	20–25	8.2	52	Toxiflow (NR)			0.017 (0.388–0.047)	McCombs, Chambers, and Yarbrough 1977

Table 3.1. Continued

Species	Test conditions*	Temperature (°C)	pH	Hardness (mg $CaCO_3$ L^{-1})	Formulation (% ai)	LC_{50} or EC_{50} (mg L^{-1}) 24 h	48 h (confidence interval)	96 h	Reference
Lepomis macrochirus (Bluegill)	S, U	12.7	7.1	ND	Toxaphene (NR)	503 (490-840)		190 (180-230)	Macek, Hutchinson, and Cope 1969
	S, U	18.3	7.1	ND	Toxaphene (NR)	360 (320-430)		120 (100-140)	
	S, U	23	7.1	ND	Toxaphene (NR)	150 (130-170)		47 (40-55)	
Salmo gairdneri (Rainbow trout)	S, U	7.2	7.1	ND	Toxaphene (NR)	230 (190-267)		152 (132-175)	Macek, Hutchinson, and Cope 1969
	S, U	12.7	7.1	ND	Toxaphene (NR)	95 (85-115)		42 (38-49)	
	S, U	3.6	7.1	ND	Toxaphene (NR)			210 (270-375)	
Lepomis macrochirus (Bluegill)	ND	29.4	ND	ND	Toxaphene (NR)	0.910	0.0784	0.0064	Cope 1965
	ND	27.4	ND	ND	Toxaphene (NR)	0.120	0.064	0.047	
	ND	18.3	ND	ND	Toxaphene (NR)	0.360	0.200	0.135	
	ND	12.8	ND	ND	Toxaphene (NR)	0.550	0.380	0.210	
	ND	18.3	ND	ND	Toxaphene (NR)	1.300	0.500	0.280	
Salmo gairdneri (Rainbow trout)	ND	ND	ND	ND	Toxaphene (NR)			0.016-0.210	Cope 1965
INVERTEBRATES									
Daphnia magna (Cladoceran) (1st instar)	S, M	23.9	7.8	272	Technical (95 %)		360 (320-1000)		Mayer and Ellersieck 1986
Daphnia pulex (Cladoceran) (1st instar)	S, M	15.0	7.8	44	Technical (95 %)		4.25 (4.0-5.3)		Mayer and Ellersieck 1986
Simocephalus serrulatus (Cladoceran) (1st instar)	S, M	15.0	7.8	44	Technical (95 %)		800 (630-1260)		Mayer and Ellersieck 1986

Table 5-4. Continued

Species	Test conditions*	Temperature (°C)	pH	Hardness (mg $CaCO_3$ L^{-1})	Formulation (% a.i.)	LC_{50} or EC_{50} (mg L^{-1}) 24 h	48 h (confidence intervals)	96 h	References
Aedes hexodontus (Larvae) (early instar)	S, M	15.0	7.4	272	Technical (95.9)	>1000		>1000	Mayer and Ellersieck 1986
Gammarus fasciatus (Amphipod) (immature)	S, M	—	7.4	44	Technical (95.9)	8700 (6300–12 200)		2300 (1400–3400)	Mayer and Ellersieck 1986
Palaemonetes kadiakensis (Shrimp) (immature)	S, M	23.0	7.4	272	Technical (95.9)	210 (162–271)		37 (26–54)	Mayer and Ellersieck 1986
Pteronarcys californica (Stonefly) (2nd year class)	S, M	15.0	7.4	44	Technical (95.9)	11000 (8400–20 000)		2800 (2100–3700)	Mayer and Ellersieck 1986
Alonella sp. (Cladoceran)	S, U	17–23	7.8	15	Emul. conc. (4%)		0.08		Nagel, Hawkins, and Nagel 1987
Daphniopsis sp. (Calanoid copepods)	S, U	17–23	7.8	15	Emul. conc. (4%)		0.08		Nagel, Hawkins, and Nagel 1987
Eucyclops sp. (Cyclopoid copepods)	S, U	17–23	7.8	15	Emul. conc. (4%)		0.05		Nagel, Hawkins, and Nagel 1987
Cypria sp. (Ostracod)	S, U	17–23	7.8	15	Emul. conc. (4%)		0.06		Nagel, Hawkins, and Nagel 1987
Simocephalus serrulatus (Cladoceran) (1st instar)	S, U	15–16	7.4–7.8	45	Technical (NR)		450 (150–620)		Sanders and Cope 1966
Daphnia pulex (Cladoceran) (1st instar)	S, U	15–16	7.4–7.8	45	Technical (NR)		280 (160–360)		Sanders and Cope 1966
Procambarus clarkii (Crayfish) (juvenile)	S, U	23–25	NR	NR	Technical (NR)	13 (12–14)		12 (11–13)	Nagel and Leong 1983
(adult)	S, U	23–27	6.8	NR	Technical (NR)			26 (23.6–28.9)	Nagel, Hawkins, and Nagel 1987
(juvenile)	S, U	23–27	6.8	NR	Technical (NR)			13 (12.1–15.0)	
Daphnia magna (Cladoceran)	S, U	21	7.4	272	Technical (NR)		0.36		Sanders 1970

Table B.4. (continued)

Species	Test conditions[a]	Temperature (°C)	pH	Hardness (mg $CaCO_3$ L^{-1})	Formulation (% ai)	LC_{50} or EC_{50} (mg L^{-1}) 24 h	48 h (confidence interval)	96 h	Reference
Cyprinodon variegatus (Cyprinodontidae)	S, U	20	7.4	272	Technical (NR)		0.25		Sanders 1970
Anabas testudineus (Anabantidae)	S, U	15.5	7.4	272	Technical (NR)		2.0		Sanders 1970
Palaemonetes kadiakensis (Palaemonidae)	S, U	20	7.4	272	Technical (NR)		1.2		Sanders 1970
Orconectes nais (Cambaridae)	S, U	15	7.4	272	Technical (NR)		50.0		Sanders 1970
Gammarus fasciatus (Amphipoda)	S, U	15.5	7.4	272	Technical (NR)	3.2 (1.9–4.7)	1.8 (1.0–3.2)	1.0 (0.5–1.6)	Sanders 1970
Pteronarcys californica (Pteronarcyidae)	S, U	21	ND	ND	ND		4.0		Cope 1966
Daphnia pulex (Cladocera)	S, U	21	ND	ND	ND		0.24		Cope 1966
Simocephalus serrulatus (Cladocera)	S, U	21	ND	ND	ND		0.47		Cope 1966
Rhithrogena robusta (Heptagenidae) (mortality)	S, W	ND (Sal = 20 g L^{-1})			Technical (NR)		0.12 (mortality)	0.10 (mortality)	Lee and Lee 1979
(adults)	S, W	ND (Sal = 20 g L^{-1})			Technical (NR)			0.28 (mortality)	
Daphnia magna (Cladocera) (<24 h old)	S, W	19–21	6.8–7.2	36–39	Technical (NR)		0.103 (0.115–0.327)		Marsh *et al.* 1979

Appendix F

Chronic Toxicity of Trifluralin to Aquatic Organisms

Table 9-6. Chronic Toxicity of Trifluralin to Aquatic Organisms

Species	Test conditions*	Temperature (°C)	pH	Hardness (mg L^{-1})	Salinity (g L^{-1})	Formulation (% ai)	Test duration (d)	Effect	Reference
VERTEBRATES									
Cyprinodon variegatus (Sheepshead minnow) (Zygote)	F, M	20	ND	ND		Trifluralin (ND)	28	5.5 µg L^{-1} caused common dysplasia of vertebrae; 2.7 µg L^{-1} apparently had no effect.	Couch *et al.* 1979
Cyprinodon variegatus (Sheepshead minnow) (Zygote)	F, M	20	ND	ND		Trifluralin (ND)	336	1–5 µg L^{-1} caused diffuse vertebral dysplasia in 17 of 20 fish, focal hyperostosis of vertebrae in 7 of 20 fish, and combined pituitary enlargement plus other histopathologic changes in 10 of 20 fish.	Couch, Courtney, and Ivan 1981; Couch 1984
(30 d old)	F, M	20	ND	ND		Trifluralin (ND)	360	1–5 µg L^{-1} caused diffuse vertebral dysplasia in 18 of 20 fish, focal hyperostosis of vertebrae in 11 of 20 fish, and combined pituitary enlargement plus other histopathological changes in 11 of 20 fish.	
Pimephales promelas (Fathead minnow) (44 d old)	F, M	24–25	6.6–7.2	23–39		Trifluralin (9%)	32	Incipient LC_{50} = 115 µg L^{-1} (95% confidence interval = 48–231 µg L^{-1})	Macek *et al.* 1976
(28 d old)	F, M	24–25	6.6–7.2	23–39		Trifluralin (9%)	42?	Maximum acceptable toxicant concentration (MATC) = >1.95, <5.1 µg L^{-1}	
INVERTEBRATES									
Cancer magister (Dungeness crab) (Zoeal stage)	F, M	12–14	7.50–8.1		32–34.5	Technical (95)	40	30% of larvae exposed to 220 µg L^{-1} survived to day 5; 100% mortality by day 9. 26 and 7.1 µg L^{-1} had no effect on survival in the period day 10–50.	Caldwell *et al.* 1979

ND = no reports

*Test conditions: S = static
F = flow-through
M = measured

Table F-1. Continued

Species	Test conditions*	Temperature (°C)	pH	Hardness (mg L^{-1})	Salinity (g L^{-1})	Formulation (% ai)	Test duration (d)	Effect	Reference
(Juvenile stages)	F, M	12–14	7.30–8.1		32–34.5	Technical (95)	30	Survival not affected by 500 µg L^{-1}.	
(Adults)	F, M	12–14	7.30–8.1		32–34.5	Technical (95)	85	Survival not affected by 300 µg L^{-1}.	
Limnodrilus hoffmeisteri (=40%); *Tubifex tubifex* (=10%); (Oligochaetes)	S, M	NR	NR	NR		Analytical Reference Std. (NR)	30	Survival and functioning of worms not affected by sediment conc. of 1.2 mg kg^{-1}	Karickhoff and Morris 1985
Daphnia magna (Cladocerans)	S, M	19–21	6.8–7.2	38–39		Trifluralin (98)	64	MATC = >2.4, <7.2 µg L^{-1}	Macek *et al.* 1976

Appendix G

Summary of Selected Trifluralin Phytoxicity Data

Table G-6. Summary of Selected Trifluralin Phytotoxicity Data

Species	Dosage	Response (relative to control)	Conditions	Reference
Cabbage (*Brassica oleracea*) (seeds)	1.12 kg ha^{-1}	120–198 increase in number of plants grown from seeds (none NR)	Field cultivated	Cassidy 1972
Cauliflower (*Brassica oleracea*) (seeds)	1.12 kg ha^{-1}	140 increase in number of plants grown from seeds; 900 increase in fresh weight (none NR)	Field cultivated	Cassidy 1972
Broccoli (*Brassica oleracea*) (seeds)	1.7 kg ha^{-1}	140 decrease in fresh weight of plants (none NR)	Field cultivated	Ivany and Cieslik 1973
	1.7 kg ha^{-1}	9% increase in fresh weight of plants (none NR)	Field cultivated	
Brussels sprout (*Brassica oleracea*) (seeds)	1.7 kg ha^{-1}	No effect on fresh weight of plants (none NR)	Field cultivated	Ivany and Cieslik 1973
Cauliflower (*Brassica oleracea*) (seeds)	1.7 kg ha^{-1}	21% fresh weight decrease in plants (none NR)	Field cultivated	Ivany and Cieslik 1973
Soybean (*Glycine max*) (seeds)	1.68 kg ha^{-1}	6%–23% of plants injured; plant number decreased by 6% at 37 d	Field cultivated	Fawcheim 1973
	0.84 kg ha^{-1}	No effect on plant growth at 37 d	Field cultivated	
Tomato (*Lycopersicon esculentum*) (seeds)	0.56 kg ha^{-1}	3% of plants injured and 14% decrease in fresh weight at maturity (none NR)	Field cultivated	Brown and Swingle 1977
Tomato (*Lycopersicon esculentum*) (seedlings)	1.0 kg ha^{-1}	8% increase in fruit yield by weight at harvest at 105 d	Field cultivated	Hoose 1977
Soybean (*Glycine max*) (seeds)	0.56 kg ha^{-1}	10% of mature plants injured after 120 d; 10% of seedlings injured after 21 d	Field cultivated	Hamaker 1975
Potato (*Solanum tuberosum*) (seedlings)	1.12 kg ha^{-1}	26% decrease in tubers at harvest at 3 mo	Field cultivated	Smeck 1976
Cabbage (*Brassica oleracea*) (seeds)	0.56 kg ha^{-1}	No effect on plants (none NR)	Field cultivated	Bullock and Smeck 1977
Cucumber (*Cucumis sativus*) (seeds)	0.84 kg ha^{-1}	30% of plants injured 35 d	Field cultivated	Ashley 1977a
Summer squash (*Cucurbita pepo condensa*) (seeds)	0.84 kg ha^{-1}	60% of plants injured 37 d	Field cultivated	Ashley 1977a
Field bean (*Phaseolus vulgaris*) (seeds)	0.56 kg ha^{-1}	9% decrease in plant number; 10% increase in bean yield at harvest 96 d	Field cultivated	Penner and Wicks 1971

NR = not reported

Table G-1. Continued

Species	Dosage	Response (relative to control)	Conditions	Reference
Baby lima bean (*Phaseolus lunatus*) (seeds)	1.12 kg ha^{-1}	13% of seedlings injured 28 d after exposure; all plants recovered by 77 d	Field cultivated	Beste 1975
	0.56 kg ha^{-1}	7% of mature plants injured 77 d	Field cultivated	
Pea (*Pisum sativum*) (mature plants)	0.6 kg ha^{-1}	50% fresh weight of mature plants 1 mo	Field cultivated	Harvey, Grimm, and Diersch 1972
Soybean (*Glycine max*) (seeds)	3.4 kg ha^{-1}	25% of plants injured (dose NR)	Field cultivated	Le Baron, Wilson, and Taylor 1971
Cabbage (*Brassica oleracea*) (seeds)	0.84 kg ha^{-1}	No effect on seedlings after 14 d	Field cultivated	Selleck and Sneed 1976
Grape (*Vitis* sp.) (dosage NR)	1.12 kg ha^{-1}	27% of plants injured (dose NR)	Field cultivated	Lange *et al.* 1969
Potato (*Solanum tuberosum*) (mature plants)	0.84 kg ha^{-1}	10% increase in tubers at harvest (dose NR)	Field cultivated	Murphy and Groves 1976
Tomato (*Lycopersicon esculentum*) (seedlings)	0.56 kg ha^{-1}	14% injury of plants after 33 d postemergence	Field cultivated	Beste 1976
Soybean (*Glycine max*) (seeds)	1.12 kg ha^{-1}	90% increase in dry weight of shoots (dose NR)	Field cultivated	Burnside 1968
Wild cane (*Sorghum halepe*) (seeds)	1.12 kg ha^{-1}	76% decrease in dry weight of shoots (dose NR)	Field cultivated	Burnside 1968
Slash pine (*Pinus elliottii*) (seedlings)	2.2 kg ha^{-1}	No effect on survival or dry weight after 1 d	Field cultivated	Dill and Carter 1973
Loblolly pine (*Pinus taeda*) (seedlings)	2.2 kg ha^{-1}	No effect on survival or dry weight after 1 d	Field cultivated	Dill and Carter 1973
Sorghum (*Sorghum vulgare*) (seeds)	4 mg L^{-1}	90% shoot fresh weight decrease; 97% decrease in leaf size after 1 mo	Greenhouse study	Hamowitz, Hales, and Blessenbiel 1978
Carrot (*Daucus carota sativa*) (seeds)	1.12 kg ha^{-1}	11% decrease in plant number; 34% fresh weight decrease in root (dose NR)	Field cultivated	Teal 1975
Soybean (*Glycine max*) (seeds)	0.84 kg ha^{-1}	6% of plants injured after 30 d	Field cultivated	Johnson 1971
Tomato (*Lycopersicon esculentum*) (seedlings)	0.56 kg ha^{-1}	15% plant injury after 28 d postemergence	Field cultivated	Grande and Ondarchie 1975

Table G-1. Continued

Species	Dosage	Response (relative to control)	Conditions	Reference
Soybean (*Glycine max*) (crude)	0.9 kg ha^{-1}	Plant number decrease (nondose NR) after 10 d	Field-untreated	Hamilton and Ashe (47)
Cucumber (*Cucumis sativus*) (crude)	2 mg L^{-1}	100% decrease in germination after 96 h	Greenhouse study with silica sand	Rantessine and Warren (47)
	265 mg L^{-1}	50% decrease in shoot size after 96 h	Lab study with silica sand	
	16 mg L^{-1}	50% decrease in root size after 96 h	Lab study with silica sand	
Soybean (*Glycine max*) (seeds)	1.4 mg L^{-1}	50% decrease in root size after 96 h	Lab study with silica sand	Rantessine and Warren (47)
	16 mg L^{-1}	100% decrease in germination after 96 h	Greenhouse study with silica sand	
	16 mg L^{-1}	50% decrease in shoot size after 96 h	Lab study with silica sand	
	5.8 mg L^{-1}	50% decrease in shoot size after 96 h	Greenhouse study with silica sand	
	3.5 mg L^{-1}	50% decrease in root size after 96 h	Greenhouse study with silica sand	
Cotton (*Gossypium hirsutum*) (seeds)	10 mg L^{-1}	100% decrease in germination after 96 h	Greenhouse study with silica sand	Rantessine and Warren (47)
	10 mg L^{-1}	50% decrease in shoot and root size after 96 h	Lab study with silica sand	
	1.8 mg L^{-1}	50% decrease in shoot size after 96 h	Greenhouse study with silica sand	
	2.2 mg L^{-1}	50% decrease in root size after 96 h	Greenhouse study with silica sand	
Barley (*Hordeum vulgare*) (seeds)	16 mg L^{-1}	100% decrease in germination after 96 h	Greenhouse study with silica sand	Rantessine and Warren (47)
	3.3 mg L^{-1}	50% decrease in root size after 96 h	Greenhouse study with silica sand	
	0.09 mg L^{-1}	50% decrease in root size after 96 h	Lab study with silica sand	
	3.3 mg L^{-1}	50% decrease in shoot size after 96 h	Lab study with silica sand	
Rice (*Oryza sativa*) (seeds)	0.17 mg L^{-1}	50% decrease in shoot size after 96 h	Greenhouse study with silica sand	Rantessine and Warren (47)
	0.40 mg L^{-1}	50% decrease in root size after 96 h	Lab study with silica sand	

Table G-1. Continued

Species	Dosage	Response (relative to control)	Conditions	Reference
	1 mg L^{-1}	100% decrease in germination after 96 h	Greenhouse study with silica sand	
	0.3 mg L^{-1}	50% decrease in shoot size after 96 h	Lab study with silica sand	
	0.18 mg L^{-1}	50% decrease in root size after 96 h	Greenhouse study with silica sand	
Pea (*Pisum sativum*) (seeds)	8 mg L^{-1}	50% decrease in shoot size after 96 h	Lab study with silica sand	Rennison and Warren 1971
	4 mg L^{-1}	50% decrease in root size after 96 h	Lab study with silica sand	
Cotton (*Gossypium hirsutum*) (seeds)	0.6 kg ha^{-1}	5% reduction in number of seedlings	Field cultivated	Miller, Carnes, and Carter 1983
Shortleaf pine (*Pinus echinata*) (seeds)	1.12 kg ha^{-1}	33% decrease in fresh weight (time NR)	Field cultivated	Smith 1977
Tomato (*Lycopersicon esculentum*) (mature plant)	100 mg m^{-2}	20% of plants exhibited deformed leaves and stems	Greenhouse study	Zilkah, Rosen, and Gamrasni 1977
Soybean (*Glycine max*) (seeds)	0.8 kg ha^{-1}	56% decrease in yield of seeds at harvest (time NR)	Field cultivated	McNevin and Harvey 1982
Pea (*Pisum sativum*) (seeds)	0.8 kg ha^{-1}	12% decrease in yield of seeds at harvest, 80 d	Field cultivated	McNevin and Harvey 1982
Sugar cane (*Saccharum officinarum*) (seeds)	4.48 kg ha^{-1}	5% of plants injured after 8 wk	Field cultivated	Barrow 1977
Shortleaf pine (*Pinus echinata*) (seeds)	1.1 kg ha^{-1}	51% decrease in fresh weight of seedlings (time NR)	Field cultivated	Gjerstad and Smith 1981
Slash Pine (*Pinus elliottii*) (seeds)	1.1 kg ha^{-1}	No effect on fresh weight of seedlings (time NR)	Field cultivated	Gjerstad and Smith 1981
Loblolly pine (*Pinus taeda*) (seeds)	1.1 kg ha^{-1}	83% decrease in fresh weight of seedlings (time NR)	Field cultivated	Gjerstad and Smith 1981
Potato (*Solanum tuberosum*) (mature plants)	12 kg ha^{-1}	83% decrease in number of of tubers in 8 mo	Field cultivated	Linnan 1977
Kidney bean (*Phaseolus sp.*) (seeds)	1.1 kg ha^{-1}	11% injury to seedlings at 48 d	Field cultivated	Hatfield, Wiebelt, and Swan 1978
	0.84 kg ha^{-1}	15% decrease in beans from mature plants at 115 d	Field cultivated	

Table G-4. Continued

Species	Dosage	Response (relative to control)	Conditions	Reference
Cabbage (*Brassica oleracea*) (seeds)	0.56 kg ha^{-1}	>90% injury to seedlings at 34 d postemergence; 22% decrease in yield at harvest (103 d)	Field cultivated 1978	Garfield, Warfield, and Foster
	0.84 kg ha^{-1}	22% injury to seedlings at 34 d postemergence; 6% decrease in yield at harvest (103 d)	Field cultivated	
Soup bean (*Phaseolus vulgaris*) (seeds)	3.4 kg ha^{-1}	Cellular injury observed in seedlings; vascular disruptions and swelling of stem at 15 d	Greenhouse study	Swarbreck, Bowing, and Harvey 1976
Pea (*Pisum sativum*) (seeds)	1.12 kg ha^{-1}	10% decrease in mature plants; 1% fresh weight decrease at 34 d	Field cultivation	Tweedale, Harvey, and Hagedorn 1978
	1.68 kg ha^{-1}	99% decrease in mature plants; 25% fresh weight decrease at 34 d	Field cultivation	
Soybean (*Glycine max*) (seeds)	3.35 mg L^{-1}	7% decrease in germination; 25% decrease in fresh weight; 64% decrease in root size; 74% decrease in shoot size at 7 d	Lab study, no soil	Harvey 1975
	1.35 mg L^{-1}	60% decrease in dry weight at 28 d	Greenhouse study	
	0.5 mg L^{-1}	25% root injury; 9% decrease in root dry weight; 13% decrease of plant fresh weight and dry weight at 34 d	Lab study, terrarium-type chamber with soil	Mumtz et al. 1976
	1 mg kg^{-1}	15% root injury; 20% decrease in root dry weight; 18% decrease in plant fresh weight; 14% decrease in plant dry weight at 34 d	Lab study, environmental chamber with soil	
	4 mg kg^{-1}	75% root injury; 66% decrease in root dry weight; 51% decrease in plant fresh weight; 53% decrease in plant dry weight at 34 d	Lab study, environmental chamber with soil	
	8 mg kg^{-1}	85% root injury; 76% decrease in root dry weight; 61% decrease in plant fresh weight; 65% decrease in plant dry weight at 34 d	Lab study, environmental chamber with soil	
Cotton (*Gossypium hirsutum*) (seeds)	4 mg kg^{-1}	82% root injury; 65% decrease in root dry weight; 39% decrease in plant fresh weight; 33% decrease in plant dry weight at 34 d	Lab study, environmental chamber with soil	Mumtz et al. 1976

Table 6.6. Continued

Species	Dosage	Response (relative to control)	Conditions	Reference
	1 mg kg^{-1}	39% root injury; 61% decrease in root dry weight; 17% decrease in plant fresh weight and dry weight at 34 d	Lab study, environmental chamber with soil	
	0.3 mg kg^{-1}	34% root injury; 29% decrease in root dry weight; 14% decrease in plant fresh weight; no change in dry weight at 34 d	Lab study, environmental chamber with soil	
	0.27 mg kg^{-1}	29% root injury; 9% decrease in root dry weight; 6% decrease in plant fresh weight; no change in dry weight at 34 d	Lab study, environmental chamber with soil	
Oat (*Avena sativa* cv. Garry)	4 mg g^{-1} (mg g soil^{-1})	51% decrease in root dry weight; 90% decrease in shoot dry weight at 19 d	Greenhouse study	Rochelle and Lacy 1979
	2 mg g^{-1} (mg g soil^{-1})	41% decrease in root dry weight; 63% decrease in shoot dry weight at 19 d	Greenhouse study	
	1 mg g^{-1} (mg g soil^{-1})	34% decrease in root dry weight; 44% decrease in shoot dry weight at 19 d	Greenhouse study	
	0.5 mg g^{-1} (mg g soil^{-1})	19% decrease in root dry weight; 29% decrease in shoot dry weight at 19 d	Greenhouse study	
Soybean (*Glycine max*) (resistant plants)	1 mg L^{-1}	No leaf chlorosis; 9% decrease in plant dry weight; 4% increase in root dry weight 28 d after spray application	Greenhouse study	Bellows et al. 1979
	2 mg L^{-1}	No leaf chlorosis; 36% decrease in plant dry weight; no change in root dry weight 28 d after spray application	Greenhouse study	
	4 mg L^{-1}	45% leaf chlorosis; 66% decrease in plant dry weight; 63% decrease in root dry weight 28 d after spray application	Greenhouse study	
	2.72 kg ha^{-1}	No plant injury after 14 d	Field cultivated	Marks, Tomlinson, and Chesters 1975
	1.68 kg ha^{-1}	7% plant injury after 14 d	Field cultivated	
Cotton (*Gossypium hirsutum*) (resistant plants)	0.56 kg ha^{-1}	No plant injury after 14 d	Field cultivated	Marks, Tomlinson, and Chesters 1975

Table 6-1. Continued

Species	Dosage	Response (relative to control)	Conditions	References
	1.12 kg/ha[a]	15% plant injury after 14 d	Field cultivated	
	1.68 kg/ha[a]	25% plant injury after 14 d	Field cultivated	
Soybean (*Glycine max*) seedlings	1 mg/L[b]	plant dry weight decrease (figure 10B)	Greenhouse study	Bayer and Santelmann (1975)
	2 mg/L[b]	plant dry weight decrease (figure 10B)	Greenhouse study	
Corn (*Zea mays*) seedlings	3.33 mg/L[b]	0 to 5% decrease in root growth noticed at 3 h	Lab study, no soil	Lignowski and Scott (1971)
	1.34 mg/L[b]	10% decrease in root elongation; 20% decrease in root dry weight; 21% root swelling at 6 h	Lab study, no soil	
	1.34 mg/L[b]	10% decrease in root elongation; 21% decrease in root dry weight; 50% root swelling at 3 h	Lab study, no soil	
	1.34 mg/L[b]	60% decrease in root elongation; 51% decrease in root dry weight; 93% root swelling at 1.5 h	Lab study, no soil	
Cotton (*Gossypium hirsutum*) seedlings	0.1 mg/L[b]	9% decrease in root size at 48 h postemergence; 13% decrease in root size and 67% decrease in root tissue nitrate rate at 72 h	Lab study, no soil	Rule (1973)
	0.3 mg/L[b]	22% decrease in root size at 48 h postemergence; 41% decrease in root size and 93% decrease in root tissue nitrate rate at 72 h	Lab study, no soil	
Snap bean (*Phaseolus vulgaris humilis*) seedlings	0.84 kg/ha[a]	10% injury to plants (see 10B)	Field cultivated	Ashley (1973b)
Cauliflower (*Brassica oleracea*) (see 10B)	1.1 kg/ha[a]	24% decrease in harvest (see 10B)	Field cultivated	Bullens (1972)
Lima bean (*Phaseolus limensis*) seedlings	0.84 kg/ha[a]	8% plant injury (see 10B)	Field cultivated	Oliver (1971)
Snap bean (*Phaseolus vulgaris*) seedlings	0.84 kg/ha[a]	5% plant injury at 4 wk	Field cultivated	Parr and Feringa (1973)
	500 mg/L[b]	10% decrease in transpiration at plant aging	Lab study	Smith and Richards (1964)

Table G-1. Continued

Species	Dosage	Response (relative to control)	Conditions	Reference
Pea (*Vigna sinensis*) (seeds)	0.56 kg·ha^{-1}	19% decrease in pea production at harvest (time NR)	Field cultivated	Glaze 1970
Pea (*Pisum sativum*) (seeds)	3.35 mg·L^{-1}	47% decrease in plant number; 51% decrease in shoot dry weight at 21 d	Greenhouse study	Harvey and Jacques 1977
Lettuce (*Lactuca sativa*) (seeds)	1.12 kg·ha^{-1}	32% decrease in plant number at 48 d	Field cultivated	Bradley and Hargreaves 1977
Snap bean (*Phaseolus vulgaris*) (seedlings)	3.35 mg·L^{-1}	27% decrease in secondary metabolism at 2 h	Lab study, no soil	Ashton *et al.* 1977
	16.75 mg·L^{-1}	80% in protein synthesis; 85% decrease in RNA synthesis rate; 90% decrease in photosynthesis; 70% decrease in secondary metabolism at 2 h	Lab study, no soil	
Soybean (*Glycine max*) (seedlings)	6.7 mg·L^{-1}	21% decrease in protein synthesis in hypocotyl at 6 h	Lab study, no soil	Moreland *et al.* 1969
Soybean (*Glycine max*) (seeds)	2.2 kg·ha^{-1}	16% decrease in plant size; 35% decrease in fruit yield at harvest; 74 d	Field cultivated	Banks and Santlemann 1978
Tomato (*Lycopersicon esculentum*) (seedlings)	100 mg·m^{-2}	25% leaf deformities; 25% stem deformities at 2 wk	Greenhouse study; applied as spray	Zilkah, Bocion, and Gressel 1977
Soybean (*Glycine max*) (mature plants)	0.8 kg·ha^{-1}	11% plant injury with chlorosis and necrotic lesions; 16% decrease in harvest yield at 5 wk	Field cultivated	Moomaw and Martin 1978
	1.7 kg·ha^{-1}	21% plant injury with chlorosis and necrotic lesions; 18% decrease in plant dry weight at 5 wk	Field cultivated	
Cotton (*Gossypium hirsutum*) (seeds)	0.28–84 kg·ha^{-1}	14%–17% decrease in shoot size at 15 d	Greenhouse study	Anderson, Richards, and Whitworth 1980
Cucumber (*Cucumis sativus*) (seedlings)	1–5 g·L^{-1} (applied to meristem)	35%–50% decrease in shoot size at 7 d	Greenhouse study	Barrentine and Warren 1971
	1–5 g·L^{-1} (applied to stem)	10%–25% decrease in shoot size at 7 d	Greenhouse study	
Oat (*Avena sativa*) (seedlings)	1.8–5.4 µg·g^{-1}	10%–46% decrease in plant dry weight at 18 d	Greenhouse study	Bucholtz and Lavy 1978

Table G-1. Continued

Species	Dosage	Response (relative to control)	Conditions	Reference
Cantaloupe (*Cucumis malo*) (seedlings)	3.4 kg·ha^{-1}	No effect on fruit number or fresh weight at 59 d	Field cultivated	Lange *et al.* 1968
Soybean (*Glycine max*) (seedlings)	0.335 mg·L^{-1} (seedlings dipped into solution)	48% decrease in mitochondria respiration; 71% decrease in ATP at 1 h	Lab study, no soil	Negi *et al.* 1968
Barley (*Hordeum vulgare*) (seedlings)	2.4 mg·kg^{-1} (as slow release formulation)	50% decrease in root number; 50% decrease in root size at 72 h	Greenhouse study	O'Sullivan and Prendeville 1974
Soybean (*Glycine max*) (seeds)	4.5 kg·ha^{-1}	7% decrease harvest yield (time NR)	Field cultivated	Hagood, Williams, and Bauman 1980
Watermelon (*Citrullus lanatus*) (seeds)	1.68 kg·ha^{-1}	58% plant injury at 32 d	Field cultivated	Elmstrom and Locascio 1974
Squash (*Cucurbita maxima*) (seeds)	0.5 mg·L^{-1} (dipped into solution)	41% decrease in cotyledon enzyme activity; 50% decrease in plant size at 3 d	Lab study, no soil	Ashton, Penner, and Hoffman 1968
Squash (*Cucurbita maxima*) (seeds)	5 µg·L^{-1} (dipped into solution)	27% decrease in root protein synthesis; 18% decrease in root RNA synthesis rate; stunted roots and shoots; 31% decrease in root DNA synthesis rate at 3 d	Lab study, no soil	Schultz, Funderburk, and Negi 1968
Cotton (*Gossypium hirsutum*) (seedlings)	1.68 kg·ha^{-1}	100% decrease in lateral root production; 37% decrease in shoot fresh weight; 33% decrease in root size	Greenhouse study	Jordan, Baker, and Barrentine 1978
	1.12 kg·ha^{-1}	89% decrease in lateral root production; 24% decrease in shoot fresh weight; 27% decrease in root size	Greenhouse study	
	0.84 kg·ha^{-1}	89% decrease in lateral root production; 22% decrease in shoot fresh weight; 19% decrease in root size	Greenhouse study	
	0.56 kg·ha^{-1}	8% decrease in ratio of cotyledon weight to stem weight in 12 d old seedlings; 35% decrease in lateral roots	Greenhouse study	Mitchell and Bourland 1986
Oat (*Avena sativa*) (seeds)	1.0 µg·g^{-1} soil	35% decrease in plant fresh weight at 14 d	Greenhouse study	Nyffeler *et al.* 1982
	0.5 µg·g^{-1} soil	25% decrease in plant fresh weight at 14 d	Greenhouse study	
Soybean (*Glycine max*) (seeds)	1.1 kg·ha^{-1}	54% decrease in N_2 fixation (time NR)	Field cultivated	Bollich *et al.* 1988

Table G-1. Continued

Species	Dosage	Response (relative to control)	Conditions	Reference
	1.7 kg·ha^{-1}	70% decrease in N_2 fixation (time NR)	Field cultivated	
Cotton (*Gossypium hirsutum*) (seeds)	0.84 kg·ha^{-1}	19% decrease in tap root length; 89% decrease in number of lateral roots (time NR)	Field cultivated	Cranfill and Rhodes 1987
	1.12 kg·ha^{-1}	27% decrease in tap root length; 89% decrease in number of lateral roots (time NR)	Field cultivated	
	1.68 kg·ha^{-1}	33% decrease in tap root length; 100% decrease in number of lateral roots (time NR)	Field cultivated	